CESifo WORKING PAPERS

11036 2024

March 2024

The Economics of Inequality and the Environment

Moritz A. Drupp, Ulrike Kornek, Jasper N. Meya, Lutz Sager

CESifo

Impressum:

CESifo Working Papers
ISSN 2364-1428 (electronic version)
Publisher and distributor: Munich Society for the Promotion of Economic Research - CESifo GmbH
The international platform of Ludwigs-Maximilians University's Center for Economic Studies and the ifo Institute
Poschingerstr. 5, 81679 Munich, Germany
Telephone +49 (0)89 2180-2740, Telefax +49 (0)89 2180-17845, email office@cesifo.de
Editor: Clemens Fuest
https://www.cesifo.org/en/wp
An electronic version of the paper may be downloaded
· from the SSRN website: www.SSRN.com
· from the RePEc website: www.RePEc.org
· from the CESifo website: https://www.cesifo.org/en/wp

CESifo *Working Paper No. 11036*

The Economics of Inequality and the Environment

Abstract

Environmental degradation and economic inequality are two of the defining challenges of the twenty-first century. We synthesize conceptual mechanisms that underpin inequality-environment interlinkages and take stock of the relevant empirical evidence. We propose three channels of interaction, describing, first, how the cost of environmental policy is distributed across households, second, how environmental benefits vary with household income, third, how income inequality and redistribution shape environmental outcomes. The three channels determine how both environmental quality and economic inequality matter for policy appraisal. We argue that it is crucial to consider inequality-environment interlinkages in economic research and policy design, as neither issue can be fully understood in isolation, and close by highlighting future research needs.

Moritz A. Drupp
Department of Economics
University of Hamburg / Germany
Moritz.Drupp@uni-hamburg.de

Ulrike Kornek
Department of Economics
Kiel University / Germany
kornek@economics.uni-kiel.de

Jasper N. Meya
Biodiversity Economics Group, German Centre for Integrative Biodiversity Research (iDiv), Leipzig University / Germany
jasper.meya@posteo.de

Lutz Sager
McCourt School of Public Policy, Georgetown University, Washington DC / USA
lutz.sager@georgetown.edu

March 25, 2024
We are grateful to Pier Basaglia, Björn Bos, Wolfgang Buchholz, Olof Johansson-Stenman, Arik Levinson, Linus Mattauch, Frikk Nesje, Martin Quaas, five anonymous reviewers as well as participants at the 2020 EIE Workshop in Leipzig for helpful comments, to Steven N. Durlauf and David H. Romer for editorial guidance, to Jasper Röder for research assistance and to Andrew Halliday and Alexandra Herter for language editing. M.D. was supported by the DFG under Germany's Excellence Strategy via the Ideas and Venture Fund and project No. 390683824, contribution to the CEN of Universität Hamburg. J.M. gratefully acknowledges the support of iDiv funded by the German Research Foundation (DFG–FZT 118, 202548816). The work of U.K. on this manuscript was supported by the CHIPS project, which is part of AXIS, an ERANET initiated by JPI Climate, and funded by FORMAS (SE), DLR/BMBF (DE, Grant No. 01LS1904A-B), AEI (ES) and ANR (FR) with co-funding by the European Union (Grant No. 776608).

1 Introduction

Environmental degradation and economic inequality have emerged as two of the defining challenges of the twenty-first century. Policy makers from around the world increasingly prioritize both issues in their national and global agendas, as exemplified by the United Nations Sustainable Development Goals. Matching this public interest is a resurgence in academic research, especially in economics, driven by recent advances in understanding the nature, causes and consequences of economic inequalities (Anand and Segal, 2008; Johnson and Papageorgiou, 2020) as well as of the widespread deterioration of environmental quality, such as due to climate change and biodiversity loss (Dasgupta, 2021; Nordhaus, 2019; Stern, 2007).

Research on the two topics has long been separate, thereby missing important interactions. The yellow vest movement in France is a case in point, highlighting how distributional concerns can stand in the way of implementing policies to preserve the environment, in this case a fuel tax increase (Douenne and Fabre, 2022). More recently, governments have shown heightening interest in the interdependencies between economic inequality and the environmental. Both the United States (US) and the European Union (EU) increasingly emphasize "environmental justice" and a "just transition", as exemplified by the recent Justice40 initiative in the US and the Just Transition Mechanism in the EU, both of which combine climate-friendly investment with redistributive objectives. Similar issues feature prominently in China's latest Five-Year Plan. Economic research is increasingly following suit.

This review synthesizes the growing literature on the various ways in which economic inequality and environmental quality interact in a single conceptual framework. While selected facets of this interaction have been explored in previous reviews (e.g., Fullerton, 2011; Bento, 2013; Banzhaf et al., 2019; Hsiang et al., 2019; Cain et al., 2023), we bring together research on different key channels of interaction between inequality and the environment that have thus far been consigned to separate literatures. This allows us to identify previously unrecognized feedbacks, rebound effects, and synergies.

We structure our review by asking how environmental policy design, which primarily targets changes in environmental quality, can take into account the ways in which economic inequality interacts with environmental outcomes. Our primary focus is on income inequality among individuals and households within countries, as opposed to intergenerational inequality (e.g., Groom et al., 2022), racial disparities (e.g., Banzhaf et al., 2019) or other forms of inequality. Our conceptual framework, presented in Section 2, yields three major components of the welfare change induced by an environmental policy. First, there will be non-market benefits and market-mediated effects from improved environmental quality, which we discuss in Section 3. Second, there will be costs from the policy, in particular through changes in consumer prices and incomes, which we discuss in Section 4. Third, government redistribution, both as a component of environmental policy and as a standalone policy, can alter environmental outcomes, as we discuss in Section 5.

In each of these sections, we review the theoretical underpinnings and discuss empirical evidence. The important role of different income elasticities emerges as a common theme. We synthesize the available empirical evidence and outline knowledge gaps, including delineating key income elasticities that have not yet been explored in the literature. We strive for an integrated overview that highlights the various ways in which these thematic areas intersect. In doing so, we seek to put individual research agendas into perspective, show how they can inform one another, and highlight potential avenues for future research. Although our conceptual framework is general, much of the empirical evidence we discuss is focused on climate change and air pollution, primarily because other areas have been less well-explored in the literature.

We close by drawing conclusions for research and policy. Our conceptual review aims to help researchers identify promising new research areas, inform decisions by policy makers, and provide students with an overview of the interlinkages between two topics that are increasingly featured in curricula. By summarizing the current state of the literature, we hope to provide an impetus for research that reaches across the boundaries between largely distinct literatures on economic inequality and environmental change. For example, an economist exploring environmental policy design with certain distributional objectives—generating, say, net effects that disproportionately benefit low-income households—should also consider feedback effects that undermine the environmental objective motivating the policy. Similarly, an economist using monetary estimates of the willingness-to-pay (WTP) for improved environmental quality to conduct a benefit-cost analysis should also consider how aggregate WTP changes when accounting for inequality. And an economist concerned with economic inequality may wish to consider additional inequalities from environmental change. These examples highlight the value of an integrative approach towards tackling both economic inequality and environmental change that we propose below.

2 Conceptual framework

To conceptualize the objectives of a policy maker who considers both environmental quality and economic inequality, we start from the Bergson-Samuelson understanding of social choice common in economic analysis (as reviewed in e.g. Fleurbaey, 2009), in which environmental policy appraisal is based on a social welfare function (SWF):

$$SWF = \Phi(..., U_i, ...),$$

where U_i is utility of individual i. An environmental policy instrument should be introduced if the change in social welfare

$$\Delta SWF = \sum_i \Phi'(U_i) \cdot \Delta U_i$$

is positive and larger than that for all alternative policies. The first component, $\Phi'(U_i)$, describes how society weights utility changes accruing to different individuals. The second component, ΔU_i, is the change in well-being experienced by individuals. Individual well-being depends on the consumption of goods and services, leisure, savings and environmental quality. Changes in well-being can be represented through changes in indirect utility, $V_i(\mathbf{p}, w, r, T_i, E_i, K_i)$, where $\mathbf{p}$ is the vector of prices of goods and services (j), w is wage rate, r is the return on capital, T_i are net governmental transfers and E_i is environmental quality, which is the same for all individuals in the case of pure public environmental goods ($E_i = E \, \forall \, i$). Finally, each individual i is endowed with capital stock K_i.

A change in environmental policy may affect all five variable components of indirect utility, and so we represent each policy as a set of changes in prices, wages and rental rates, transfers as well as environmental quality $\{\Delta \mathbf{p}, \Delta w, \Delta r, \Delta T_i, \Delta E_i\}$. Using the envelope theorem, the first-order change in indirect utility of individual i under small policy-induced changes can be approximately disentangled into three parts:

$$\begin{aligned} \Delta V_i &\approx \frac{\partial V_i}{\partial E_i} \cdot \Delta E_i + \sum_j \left(\frac{\partial V_i}{\partial p_j} \cdot \Delta p_j \right) + \frac{\partial V_i}{\partial w} \cdot \Delta w + \frac{\partial V_i}{\partial r} \cdot \Delta r + \frac{\partial V_i}{\partial T_i} \cdot \Delta T_i \\ &= \frac{\partial V_i}{\partial T_i} \Bigg[\underbrace{\frac{\partial V_i}{\partial E_i} / \frac{\partial V_i}{\partial T_i} \cdot \Delta E_i}_{\text{Non-market env. benefits}} + \underbrace{\sum_j -C_{ij} \cdot \Delta p_j}_{\text{Price changes}} + \underbrace{\Delta Y_i}_{\text{Income changes}} \Bigg], \end{aligned} \tag{1}$$

where we have used Roy's identity for individual i's consumption of good j: $C_{ij} = -\frac{\partial V_i}{\partial p_j} / \frac{\partial V_i}{\partial T}$. Individual i's income change, ΔY_i, is itself the sum of changes in wage, capital and transfer income ($\Delta Y_i = \Delta w \cdot L_i + \Delta r \cdot K_i + \Delta T_i$) which in turn depend on labor supply $L_i = \frac{\partial V_i}{\partial w} / \frac{\partial V_i}{\partial T}$ and capital $K_i = \frac{\partial V_i}{\partial r} / \frac{\partial V_i}{\partial T}$, using the envelope theorem.[1]

The change in well-being in Eq. (1) is first-order and takes general equilibrium effects into account where prices for goods and services, factor prices and environmental quality change marginally. We use it to structure our review. In Sections 3 - 4, we discuss further general equilibrium effects where prices, demand and supply of goods and services as well as input factors adjust non-marginally to an environmental policy (e.g., Känzig, 2021) and consider individual-specific prices, such as heterogeneous wage rates.

Eq. (1) depicts three additive effects on indirect utility. First, there are direct benefits from the change in environmental quality induced by a policy (first term). Second, there are price changes (second term), which can occur for a number of reasons. Most immediately, prices of targeted goods (usually polluting ones) change, either because price instruments directly apply to them or because regulation makes them more expensive to produce. These are commonly denoted as "use-side" policy costs. Further price changes

[1] We represent i's choices by reducing the intertemporal setting in Goulder et al. (2019) to a two-period framework: $V_i(\mathbf{p}, w, r, T_i, E_i, K_i) = \max_{\mathbf{C}_i, L_i, S_i} U_i(\mathbf{C}_i, \bar{L} - L_i, S_i, E_i)$, s.t. $\mathbf{p} \cdot \mathbf{C}_i = w \cdot L_i + r \cdot K_i + K_i - S_i + T_i$, where $\mathbf{C}_i$ is the $\mathbf{J}$-vector of consuming goods, K_i the capital stock and S_i savings for the next period. Consumption smoothing motives in this two-period framework are represented through savings entering the utility of households.

may arise because production costs depend on environmental quality which has improved. In addition, some policy instruments generate revenue which can be used to lower goods prices, usually through tax cuts or subsidies. We use this distinction to guide the exposition and interpretation, dividing price changes into three channels—policy-induced price changes, market-mediated environmental quality effects, and redistribution of revenue: $\Delta p_j = \Delta p_j^{\mathrm{P}} + \Delta p_j^{\mathrm{E}} + \Delta p_j^{\mathrm{R}}$.[2] For example, congestion or fuel pricing will raise the cost of driving downtown (Δp_j^{P}) and affect house prices in areas with improved air quality (Δp_j^{E}), and the collected revenue may be used to grant rebates or tax breaks (Δp_j^{R}) to commuters.

Finally, there are income changes (third term), which we split into three similar effects: $\Delta y_i = \Delta y_i^{\mathrm{P}} + \Delta y_i^{\mathrm{E}} + \Delta y_i^{\mathrm{R}}$. First, incomes can change as a direct consequence of the environmental policy (P) affecting returns to production factors. These are commonly denoted as "source-side" policy costs. Second, incomes can be affected by changes in environmental quality (E) due to the policy. Third, revenue recycling can act as income redistribution. For example, a carbon tax may reduce the incomes of workers in the energy sector (Δy_i^{P}) while reduced climate damages raise agricultural productivity and thus the income of farm workers (Δy_i^{E}). In addition, the tax revenue can be paid out to households (Δy_i^{R}), for instance as a "carbon dividend".

Introducing these distinctions and rearranging gives the individual contribution to the change in social welfare ($\Phi'(V_i) \cdot \Delta V_i$) :

$$\Delta SWF_i \approx \Bigg[\underbrace{\underbrace{\frac{\partial V_i}{\partial E} / \frac{\partial V_i}{\partial y} \cdot \Delta E_i}_{\text{Non-market}} + \underbrace{\left(-\mathbf{C}_i \cdot \Delta \mathbf{p}^{\mathrm{E}} - \Delta y_i^{\mathrm{E}}\right)}_{\text{Market-mediated}}}_{\text{Environmental benefits}} \tag{2a}$$

$$+ \underbrace{\underbrace{\left(-\mathbf{C}_i \cdot \Delta \mathbf{p}^{\mathrm{P}}\right)}_{\text{Use-side}} + \underbrace{\Delta y_i^{\mathrm{P}}}_{\text{Source-side}}}_{\text{Environmental policy costs}} \tag{2b}$$

$$\underbrace{+ \left(-\mathbf{C}_i \cdot \Delta \mathbf{p}^{\mathrm{R}}\right) + \Delta y_i^{\mathrm{R}}}_{\text{Revenue recycling (where available)}} \Bigg] \underbrace{\Phi'(V_i) \cdot \frac{\partial V_i}{\partial y}}_{\text{Distributional weights}} . \tag{2c}$$

To illustrate, consider the effects of a carbon tax on a household whose head is working as a self-employed building contractor i. First, the policy will improve climate conditions and generate environmental benefits. These can include direct benefits to the contractor from reduced climate damages (ΔE_i), as well as market-mediated effects from changes in the price of climate-sensitive goods, such as cheaper agricultural products that the contractor consumes ($\Delta \mathbf{p}^{\mathrm{E}}$), and changes in the ability to generate income, such

[2]For larger changes, interaction effects make the split into three elements less clear-cut.

as increased labor productivity of the contractor (Δy_i^{E}). Second, the environmental policy will have costs. In particular, the carbon tax will raise prices of polluting goods that the contractor consumes ($\Delta \mathbf{p}^{\mathrm{P}}$), and it could alter the contractor's income by raising the cost of material or fuel inputs (Δy_i^{P}), which he may not be able to fully pass-through. Third, the contractor may benefit if the government uses tax revenues, for instance, to subsidize public transport ($\Delta \mathbf{p}^{\mathrm{R}}$) or lower income taxes (Δy_i^{R}). Finally, the change in the contractor's utility will enter social welfare through weights ($\Phi'(V_i) \cdot \frac{\partial V_i}{\partial y}$), where $\Phi'(V_i)$ is the weight of individual i in the SWF, sometimes referred to as 'social weight' , while $\frac{\partial V_i}{\partial y}$ is the marginal utility of income of individual i at the status quo.

The choice of these weights relates to fundamental welfare economic considerations recently reviewed for environmental policy appraisal by Adler (2016) as well as Fleurbaey and Abi-Rafeh (2016). For a utilitarian SWF, all individuals have the same social weight $\Phi'(V_i) = 1$, but are weighted by their marginal utility of income. A common generalization is an isoelastic, additive separable SWF with a single parameter for society's inequality aversion regarding the distribution of individuals' utility (see e.g., Adler 2016): $SWF = \sum_i^n \frac{V_i^{1-\rho}}{1-\rho}$ with $V_i > 0$ as a positive representation of individual utility. Here $\rho \geq 0$ measures society's aversion to inequality, so that $\Phi'(V_i) = V_i^{-\rho}$. The larger society's inequality aversion ρ, the more weight is given to the worse-off. Inserting income as a money-metric measure of utility, $V_i = y_i$, Atkinson (1970) obtained the constant relative inequality-aversion SWF under certain normative assumptions about distributive justice (see also Del Campo et al. (2024) and Johansson-Stenman 2005). Much of the literature tends to ignore distributional weights, as does environmental policy appraisal in practice, thereby implicitly assuming a utilitarian welfare function with constant marginal utility of income.

Applied policy appraisal frequently classifies distributional effects as either "regressive" or "progressive". To do so, households are stratified based on a welfare measure, usually annual income or expenditure (Parry et al., 2006). Incidence analysis assesses the relative welfare change for each household. A policy where the costs as a share of income (or expenditure) fall disproportionately on the less well-off is termed *regressive* and tends to increase inequality. The costs of a *progressive* policy instead fall disproportionately on the better-off and reduces inequality. On the benefits side, we define the effects as regressive (progressive) if benefits disproportionately accrue to the better-off (worse-off) individuals.[3]

Our classification of the principal effects of an environmental policy on individual welfare motivates the following sections and subsections of our review. First, in Section 3, we discuss how the benefits from environmental improvements induced by a policy are distributed. Subsections discuss separately the literature on direct non-market benefits and that on market-mediated ones. Second, in Section 4, we discuss how the

[3]While the technical definition of "progressive" ("regressive") refers to a cost or benefit that rises faster (slower) than income, we here use the common terminology of "progressive' ("regressive") capturing effects that disproportionately benefit the poor (rich). Thus, a progressive distribution of benefits and a progressive distribution of costs are both "pro-poor".

costs of an environmental policy instrument are distributed across income levels. In subsections, we assess both the longstanding literature on the incidence of costs from higher prices ("use-side") and the more recent literature on the costs from changing factor incomes ("source-side"). Third, Section 5 considers redistributive policies. Environmental policy proposals are oftentimes accompanied by complementary redistributive measures in the form of revenue recycling, as we discuss in the first subsection. But even stand-alone redistribution may have considerable repercussions for the environment, as we discuss in the second subsection.

3 Incidence of environmental benefits

This section investigates how environmental benefits resulting from environmental policy are distributed. First, we consider non-market environmental benefits as a direct source of utility. Second, we examine how environmental benefits are mediated by markets to alter the prices of goods and the returns to production factors and thus incomes.

3.1 Non-market environmental benefits

We start by isolating how a marginal change in environmental quality ΔE_i translates into indirect utility changes along the income distribution, as captured by the term $\frac{\partial V_i}{\partial E_i} / \frac{\partial V_i}{\partial y_i} \cdot \Delta E_i$ in Eq. (2b).

First, we consider changes in the provision of an environmental public good that is consumed equally by everyone, $\Delta E_i = \Delta E$. And example of this might be non-use services derived from biodiversity, like existence values. Assuming that individuals differ in income, y_i, but not in their preferences, distributional effects are determined by differences in the valuation of environmental quality—commonly assessed using the WTP for a marginal change in E—across the income distribution (Ebert, 2003; Baumgärtner et al., 2017). For a standard constant-elasticity-of-substitution (CES) utility function, with an environmental good, E, and a numeraire consumption good, $C_i = y_i$, individual marginal WTP for the environmental improvement can be approximated as:

$$WTP_i = \kappa\, y_i^{\eta^W} \qquad \text{with} \qquad \kappa := \frac{1-\alpha}{\alpha} E^{-\eta^W}, \tag{3}$$

where α is the utility share parameter for the environmental public good, and $\eta^W := \frac{\partial WTP}{\partial y} \frac{y_i}{WTP_i}$ is the *income elasticity of WTP*, which is inversely related to the CES (cf., Ebert, 2003; Baumgärtner et al., 2017; Smith, 2023).[4]

The *income elasticity of (marginal) WTP* indicates the distributional effect of the benefits of a pure public good. For instance, when $\eta^W < 1$, WTP for a marginal increase

[4]Note that this is derived from a "pseudo-choice problem" (Ebert, 2003) describing an individual's WTP for a hypothetical environmental improvement, as the level of environmental quality cannot be individually chosen in the case of rationed public goods. As such, the income elasticity of WTP for the environmental good at the rationed quantity also differs from the income elasticity of demand (Flores and Carson, 1997).

in the environmental public good increases by less than one percent when income increases by one percent. In that case, lower-income households have a higher WTP for a marginal increase in public goods relative to their income and environmental benefits are distributed progressively, i.e. pro-poor (cf., Ebert, 2003). If, in contrast, WTP increases more than proportionally with income ($\eta^W > 1$), benefits are distributed regressively, i.e. pro-rich.

Most empirical studies estimate constant income elasticities of WTP for environmental benefits that are smaller than unity (e.g., Kristrom and Riera, 1996; Jacobsen and Hanley, 2009; Drupp, 2018; Drupp and Hänsel, 2021), implying that environmental benefits are distributed pro-poor. Jacobsen and Hanley (2009), for instance, estimate the income elasticity of WTP for biodiversity conservation as 0.38 based on 145 WTP-income pairs from 45 contingent valuation studies. Drupp et al. (2024) provide a meta-analysis encompassing all types of non-market environmental goods, drawing on 851 income-WTP pairs from 397 contingent valuation studies. Using clustered robust regression, with a host of study-level controls, they find an income elasticity of WTP of around 0.79 (95-CI: 0.60 to 0.97). The mean income elasticity is lowest for recreational services (0.69) and highest for water regulation (0.85). Income elasticities do not differ significantly across key domains of environmental quality (climate regulation, air and water quality or biodiversity preservation) and, while largely situated in the pro-poor domain, 95 confidence intervals of income elasticities frequently overlap with unity (see Figure 1).

Some first research explores how income elasticities of WTP vary along socio-economic contexts, such as income. Barbier et al. (2017), for instance, estimate that the income elasticity of the WTP for water quality improvement in the Baltic sea is only 0.1–0.2 for low-income respondents, while it is 0.6–0.7 for high-income respondents. So far, there is limited theory to inform the shape of non-constant elasticities (Barbier et al., 2017; Drupp, 2018), and the cross-country meta-analysis by Drupp et al. (2024) does not find that income elasticities differ systematically across income levels. Overall, the evidence to date suggests that environmental benefits tend to be distributed pro-poor.

Second, we consider changes to environmental goods, ΔE_i, that are heterogeneously distributed across individuals, such as the health benefits due to more ambient climate or improved air quality (e.g., Carleton et al., 2022; Cohen and Dechezleprêtre, 2022). Now, we also have to consider how changes in environmental exposure vary along the income distribution. In many instances, ΔE_i is a function of the distance of an individual i to an environmental (dis)amenity, such as an urban green park or a source of pollution.Keeping with our focus on income elasticities, the empirical relationships between environmental exposure and income could also be conceptualized an *income elasticity of environmental exposure*.

Furthermore, we have to be mindful of a host of mediating factors that correlate with income, such as adaptive behavior, sorting or siting decisions by firms, which render the correlation between access to environmental goods and income endogenous (e.g., Banzhaf et al., 2019; Carleton et al., 2022; Hsiang et al., 2019). There is an empirical literature investigating how adaptive behavior depends on income, mainly looking at defensive expenditures, avoidance behavior as well as sorting and migration. Concerning defensive expenditures, Sun et al. (2017), for instance, show higher income households in China are more likely to invest in expensive air filters. While they only compare effects across three income groups, their results tentatively suggests that defensive expenditures increase inequality in exposure to air pollution along the income distribution. Concerning avoidance behavior, Chen et al. (2020) examine the impact of air pollution on short-term aviation trips in China and show that the number of passengers on the flight increases with the amount of air pollution in the origin city relative to the destination city. The number of first-class passengers increases about three times faster than the number of economy-class passengers, providing some indication of differential avoidance behavior along the income distribution. Furthermore, Zivin et al. (2011) compare changes to bottled water purchase following water quality violations. Comparing the lowest and the top income quartiles, they find no significant differences for short-term violations related to microorganisms and nitrates; however, richer households respond by buying relatively more bottled water when faced with violations that lead to longer-term health risks, such as those related to levels of chemicals.

While avoidance behavior typically focuses on the shorter term, sorting and migration occur in response to more persistent changes in the spatial distribution of environmental quality. The sorting literature à la Tiebout considers households that trade-off higher housing costs with higher amenities and "vote with their feet" in sorting towards neighborhoods with a desired bundle of taxes and local public goods, including environmental amenities (Banzhaf and Walsh, 2008; Depro et al., 2015; Banzhaf et al., 2019; Chen et al., 2022). Both the desire to sort, expressed in terms of WTP for a cleaner environment, and the ability to sort depend on income; as is also often the case for mediating factors, such as information access (e.g., Hausman and Stolper, 2021; Gao et al., 2023).[5] The theoretical prediction is that, ceteris paribus, higher income households will sort into neighborhoods with better environmental amenities (e.g., Brueckner et al., 1999; Lee and Lin, 2018). Lee and Lin (2018) show that persistent natural amenities attract high-income households, an effect which may be denoted as "coming to the amenity". The opposite effect is commonly termed "coming to the nuisance" effect (e.g., Depro et al., 2015).[6] Sorting can thus lead to an increase in environmental inequality along the income distribution.

Because adaptive behavior tends to be a function of income, it is again helpful to conceptualize it with an *income elasticity of adaptation* when exploring distributional effects. Gilli et al. (2024), for instance, estimate the income elasticity of climate damages and contrast a case with and without adaptation, using the climate impact functions by Burke et al. (2015) and data on income deciles at the country-level. They estimate mean income elasticities of climate damages of 0.76 before and 0.64 after adaptation, with 95-

[5]The Environmental Justice literature also explores related effects due to (racial) discrimination (e.g. Banzhaf et al., 2019; Christensen and Timmins, 2022).

[6]The nuisance may also come disproportionally to disadvantaged communities via siting decisions of firms (e.g., Wolverton, 2009), be relocated from such communities (Wang et al., 2021), or be remedied less often after a firm's wrongdoing via in-kind settlements (Campa and Muehlenbachs, 2023).

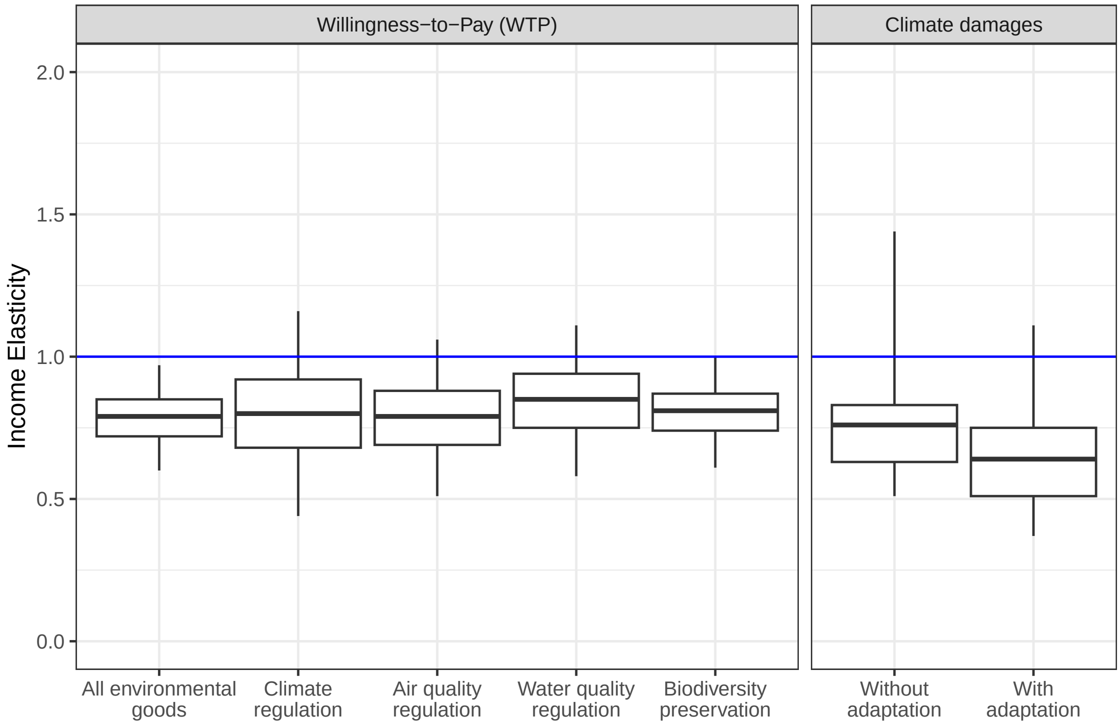

Figure 1: Estimates of income elasticities of willingness to pay (WTP) for overall and selected environmental goods based on a meta-analysis of the contingent valuation literature from Drupp et al. (2024) (left panel), and estimates of the income elasticity of climate damages from Gilli et al. (2024) (right panel), with mean estimates (middle lines), interquartile ranges (boxes) and 95 confidence intervals (whiskers). Estimates below (above) the unity-line imply that environmental benefits are distributed pro-poor (pro-rich), while the reverse holds for the incidence of climate damages.

confidence intervals that overlap unity (see Figure 1). The results suggest that climate damages tend to hit the poor disproportionally and that adaptation may exacerbate the regressivity damages from climate change.

In sum, the incidence of non-market environmental benefits is shaped by the *income elasticity of WTP*, which can it itself depend on how *environmental exposure* and *adaptation* vary with income.

3.2 Market-mediated environmental benefits

Besides leading to direct utility impacts, environmental improvements can also have effects that are mediated by markets, such as altering good prices or incomes (represented by $\Delta \mathbf{p}^{\mathrm{E}}$ and Δy_i^{E} in Eq. (2b)). Let us first consider the effect of a policy that improves local air quality. All else equal, this raises the utility of residents in the local area (see above discussion). But it also renders neighborhoods more attractive, driving up rental prices, which is an extra cost to renters but a benefit to homeowners (e.g., Grainger, 2012; Bento et al., 2015). Typically, this benefits rich owners disproportionately, but can vary by local context. Bento et al. (2015), for instance, examine the distribution of benefits of the 1990 Clean Air Act Amendments in the US. They find that, due to heterogeneous policy exposure which targeted dirtiest areas for cleanup, house price appreciation following air quality improvements disproportionately favored households in the lowest quintile of the income distribution. Such nuanced effects also explains why effects of "environmental gentrification" (cf., Banzhaf et al., 2019), which may follow changes in environmental quality, depend on pre-existing ownership and rental constellations.[7] Further knock-on effects from better air quality might include restaurants charging higher prices for serving guests outdoors or outdoor workers becoming more productive and thus collecting higher wages (Aguilar-Gomez et al., 2022).

Next, let us consider an environmental policy that reduces greenhouse gas emissions and thus mitigates climate change. Reducing the occurrence of excessive heat may raise human capital formation and labor productivity (e.g., Park et al., 2020; Somanathan et al., 2021; De Lima et al., 2021), raising wage incomes as well as lowering capital depreciation, Δy_i^E. For example, Dillender (2021) uses fixed-effects models to show that temperature extremes increase occupational injury rates in Texas as well as in the mining industry throughout the US. High temperatures have more severe adverse effects in warmer climates suggesting limited potential of avoidance behavior as an adaptation strategy for outside workers. Relatedly, Park et al. (2021) draw on injury claims between 2001 and 2018 from the US's largest worker's compensation system to explore the relationship between temperature and workplace safety. They find that the effect of heat days (with temperatures exceeding 90 degrees Fahrenheit) on work injuries tends to be distributed regressively. Similarly, climate policy can reduce crop yield losses (e.g., Ortiz-Bobea et al., 2021; Schlenker and Roberts, 2009). Thus, improved environmental quality can lead to higher incomes for farmers or lower prices of agricultural goods.

Overall, the incidence of the price changes due to the environmental policy may be summarized by the *income elasticity of demand for environmentally exposed goods*, η^{C_E}, and the incidence of income changes by the *income elasticity of income from environmentally exposed factors*, η^{F_E}, including effects on both labor and capital incomes. Consider the case in which improved environmental quality leads to lower prices for certain agricultural products (e.g. due to fewer crop failures). Since agricultural products are—as a share of the total budget—consumed disproportionately by lower-income households, the *income elasticity of demand for environmentally exposed goods*, η^{C_E}, would in this case be lower than unity and market-mediated environmental benefits of the environmental policy would benefit poorer individuals disproportionately.

4 Incidence of environmental policy costs

This section considers how environmental policy costs are distributed and thus affect inequality, extending prior reviews (Fullerton, 2011; Bento, 2013) by synthesizing recent

[7] The literature also highlights the role of non-monetary barriers, such as (racial) discrimination (e.g., Christensen and Timmins, 2022; Christensen et al., 2021).

theoretical and empirical contributions. We focus on taxes, standards, and permits that primarily target climate change and air pollution as we find that most literature assesses how the costs of energy-related environmental policy are distributed.

The incidence of environmental policy costs is represented by $-\mathbf{C}_i \cdot \Delta \mathbf{p}^{\mathrm{P}} + \Delta y_i^{\mathrm{P}}$ in Eq. (2a). Concerning the first component, individuals are affected through changes in relative prices of goods and services ("use-side incidence"); concerning the second, they are affected through changes in their factor incomes ("source-side incidence"). In the former effect, consumers are directly affected. In the latter effect, producers are affected, which indirectly affects consumers through their income. As a basic insight from general equilibrium theory, the economic costs of a tax are generally shared between producers and consumers, irrespective of whether consumers or firms physically pay the tax (Weyl and Fabinger, 2013). If a policy directly applies to consumers, for example subsidies for public transport, consumers adjust their demand and firms adjust their supply, resulting in an overall shift of prices and income. If policies target firms and increase their cost of producing, for example a carbon tax, firms pass on part of the extra costs to consumers. Direct compliance costs of an environmental policy are then partly passed on to consumer prices, while the rest is retained by producers. For the case of a tax increase in a perfect market, Weyl and Fabinger (2013) show that consumers bear a share ρ and producers bear a share $1 - \rho$ of the extra policy costs, where ρ is the pass-through rate of the tax to consumer prices. In their model, consumers have a negative price elasticity of dirty demand ϵ^D and producers a price elasticity of dirty supply ϵ_S. Without market distortions, the pass through rate of a tax to consumers is $\rho = \frac{\epsilon^S}{\epsilon^S - \epsilon^D}$. Hence the more inelastic part bears the bigger burden.

Empirical estimates have often found pass-through rates to consumers of close to 100%: Li et al. (2014) for taxes on gasoline in the US, Andersson (2019) for energy and carbon taxes on gasoline in Sweden, and Fabra and Reguant (2014) for emission taxes on electricity production in Spain. A recent body of literature, however, highlights deviations from these pass-through rates. First, pass-through rates depend on socio-economic variables. Harju et al. (2022) study a large increase in the Finnish carbon tax on fuel, reporting an average pass-through rate of only 80%, which varies between 76 and 91% depending on income and urban/rural divide of retail locations, with lower-income or more rural areas facing higher pass-through rates. They discuss that both geographical characteristics and attention to the tax increase may be responsible for rather low pass-through rates. Second, the scope of the regulation matters. Muehlegger and Sweeney (2022) show that idiosyncratic cost shocks to oil refineries in the US are hardly passed on, while common cost shocks to all firms, for example in the form of a comprehensive carbon tax, are fully passed on to consumers. Third, the market structure matters. Preonas (2023) shows that market power in the railroad sector may significantly decrease the pass-through rate of a carbon tax on coal fired power plants, finding a value as low as 75% for some plants. Data on US coal transportation shows that coal power plants adjust their coal demand due to changes in gas prices when gas competes with coal for electricity generation. Rail carriers reoptimize their mark-ups in response to changed coal demand from coal power plants that face imperfect competition in the transport sector. Spurlock and Fujita (2022) also find no price increase in response to stricter energy efficiency standards for clothes washers in the US, which they attribute to strategic pricing of suppliers and, possibly, innovation externalities when producers developed products to comply with the efficiency requirements. Finally, Ganapati et al. (2020) study the pass-through of energy cost shocks for 6 US manufacturing industries with a focus on intermediate goods and find that marginal cost-pass through rates often deviate strongly from unity. While the pass-through rate of marginal costs to prices is 70% on average, it exceeds 100% for some industries, which Ganapati et al. relate to imperfect competition. They further show how imperfect competition alters the split between consumer and producer share in welfare losses due a tax.

We now turn to reviewing the literature on how environmental policy costs differ by income group. Studies can be distinguished with respect to which economic adjustments in response to the policy change they take into account. Building on Eq. (2a), we first review how policy costs affect individuals through use-side effects. Second, we examine how policy costs change individual income through source-side effects.

4.1 Use-side effects

The use-side effect is represented by $\left(-\Delta \mathbf{p}^{\mathrm{P}} \cdot \mathbf{C}_i\right)$ in Eq. (2b). The literature assessing use-side effects commonly relies on exogenously fixed pass-through rates of policy costs to consumers to calculate the price change of goods and services $\Delta \mathbf{p}^{\mathrm{P}}$, and often assumes that consumers bear 100% of the additional costs from the policy (e.g., Dorband et al., 2019; Cronin et al., 2019). Fixing a pass-through rate, price changes due to the environmental policy should be calculated for all goods and services to include direct and indirect price changes. Direct price changes capture household consumption that generates pollution (e.g. burning fossil fuels for heating). Indirect price changes capture goods and services that contain dirty intermediate inputs (e.g. electricity to produce electronic devices). Using input-output analysis to calculate how carbon taxes affect prices of all goods and services, Cronin et al. (2019) and Feindt et al. (2021) show that indirect price effects can be important and can sometimes even overturn the distributional incidence of direct effects. The meta-analysis in Ohlendorf et al. (2021) collects evidence from 53 empirical studies in 39 countries and indicates a higher likelihood of a progressive incidence of carbon pricing for studies that include indirect effects.

As environmental policy targets environmentally harmful goods, the use-side incidence is generally driven by relative price increases of dirty consumption. Denoting total demand for dirty goods D_i and the associated price change $\Delta p^{\mathrm{D}} > 0$, the use-side effect $\left(-\mathbf{C}_i \cdot \Delta \mathbf{p}^{\mathrm{P}}\right)$ in Eq. (2a) becomes $\left(-D_i \cdot \Delta p^{\mathrm{d}}\right)$ when clean consumption is the numeraire (i.e. the price change of clean consumption is zero). When prices increase and demand is inelastic, the distribution of this burden across income groups follows the *income elasticity of dirty demand*: $\eta^D = \mathrm{d}\ln(D_i)/\mathrm{d}\ln(y_i)$. The burden is regressive (i.e. pro-rich) if η^D is smaller than 1: budget shares of dirty consumption determine the bur-

Table 1: Annual income, expenditure and price elasticities of selected goods

Good	Income elasticity of demand	Expenditure elasticity of demand	Price elasticity of demand
Electricity	0.05 (USA)[a] 0.1 (China)[b]	0.4 (Germany)[c] 0.6 (China)[d]	-0.3 (short term) -0.6 (long term)[e]
Water	0.1 (USA/Canada)[f] 0.1 (Vietnam)[g]	0.1 (Spain)[h] 0.2 (Cambodia)[i]	-0.4[j]
Gasoline	0.2-0.3 (USA)[k] 1.2 (Mexico)[l]	0.3 (USA)[m] 1.2 (China)[n]	-0.2 (short term) -0.8 (long term)[o]
Beef	1.0 (Sweden)[p] 0.4 (China)[q]	1.5 (Germany)[r] 1.9 (Malaysia)[s]	-0.9[t]

For references *a-t*, see online Appendix.

den of the policy as a share of income $\Delta p^d d_i / y_i \propto y_i^{\eta^D - 1}$, which decreases in income if $\eta^D < 1$. Likewise, the burden is progressive (i.e. pro-poor) if η^D is larger than 1.

Table 1 reports income elasticities of demand for specific goods. The reported goods can be expected to experience an increase in their relative price due to environmental regulations for, e.g., CO_2 emissions, sulfur emissions, pesticides or water contamination. Due to the lack of systematic evidence, selective studies per good are reported from diverse countries. For the selective studies, the income elasticities of electricity and water are consistently below one, contributing to a regressive use-side incidence particularly for water. The picture is less clear for gasoline and beef which show values above or below one, depending on the country.

The literature on policy incidence analysis has stressed the importance of the welfare measure by which households are compared, usually either annual income or expenditure. Poterba (1989) argues that incidence analysis should be based on expenditure rather than income because expenditure is a proxy for lifetime income based on the permanent income hypothesis (Friedman, 1957).[8] Studies frequently find that a policy which resembles an excise tax is less regressive when incidence is measured based on expenditure instead of income (Fullerton and Heutel, 2011; Cronin et al., 2019; Douenne, 2020). This finding was confirmed in a meta-analysis of carbon pricing: Ohlendorf et al. (2021) find that using proxies for lifetime income instead of annual income increases the likelihood of a progressive incidence. Table 1 therefore also reports expenditure elasticities. The expenditure elasticities are never below the income elasticities, confirming that the incidence based on expenditure tends to be less regressive than based on income. Especially beef is a luxury good when using expenditure.

[8]Based on this insight, Hassett et al. (2009) construct two proxies for lifetime income. The first is current expenditure, which corrects for smoothing shocks to current income anticipating lifetime income. The second corrects for cycles in consumption patterns over time. Here, a specific age-group might have a high share of dirty goods in their current expenditure, but over the entire lifetime the share might be rather proportional. The second measure corrects for a potential bias due to life-cycle consumption patterns.

If demand is price elastic, consumption levels adjust and welfare changes go beyond the product of price changes and budget shares and include further economic adjustments. Including behavioral responses, welfare changes should be calculated as the equivalent variation, the compensating variation or approximations such as the change in consumer surplus. These indicators can include behavioral change at the intensive margin, such as in the form of consuming less of a good, and at the extensive margin, such as in the form of switching to a different technology. The most comprehensive studies of demand adjustments include leisure as a good, taking into account that more or less time may be spent working (Goulder et al., 2019). Demand adjustments reduce the policy burden when households avoid polluting behavior. West and Williams III (2004) compute the incidence of a gasoline tax in the US based on four measures of welfare change: equivalent variation (1), consumer surplus change under demand adjustments that are either income specific (2) or not (3) and assuming inelastic demand (4). They show that demand adjustments reduce the regressivity of a gasoline tax, an effect that is also present in the meta-analysis of Ohlendorf et al. (2021) for carbon pricing.

Price elasticities of demand represent behavioral changes, which are estimated via demand systems. The influence of price elasticities on the use-side incidence has two important components. First, elasticities differ by product type. Low-income households will particularly suffer from an environmental policy that targets a good that they inelastically demand and disproportionately consume. Table 2 reports price elasticities of demand based on meta-analyses for specific goods. Demand is inelastic to price increases for all goods, especially in the short term. We note two important caveats to the reported low average absolute price elasticities: First, recent research finds stronger demand responses to price changes from taxes than from other sources (Li et al., 2014; Andersson, 2019; Basaglia et al., 2024). Second, meta-analyses report a large variation of elasticity estimates depending on methods of analyses or geographical location. Comparing countries at different development stages, households in lower income countries are more responsive to higher energy prices in the long-term (Labandeira et al., 2017) and to higher food prices (Green et al., 2013; Femenia et al., 2019).

Second, for distributional analysis it is important to know whether different income groups react differently to price changes. There is limited systematic evidence and more research needed on the pattern between income and price elasticities of demand for different goods and different countries. We report some conflicting evidence. The meta-analysis of Green et al. (2013) finds that lower-income groups tend to respond more elastically to food prices than higher-income groups. However, results again depend on the specific good and socio-economic context: Kumar et al. (2011) find that higher-income groups react more elastically to price increases for vegetables, fruit and milk in India. For energy prices, no systematic evidence exists: in some settings low-income households react more strongly to energy price changes (West and Williams III, 2004, for the US), while others report the opposite (Frondel et al., 2019 for Germany).

The discussion so far shows that the use-side incidence of an environmental policy is pollutant specific, and depends on further socio-economic characteristics such as the geography and development stage of a country. We thus now analyze incidence estimates of specific pollutants by relying on studies that provide or discuss wider evidence than one specific policy or country, where possible. We discuss the incidence of transport policies, policies on carbon emissions and local air pollutants.

Taxing transport fuels is progressive in many countries, with high-income countries showing less progressivity or a regressive effect. Sterner (2012), Chapter 19, summarizes the incidence of fuel taxes in 22 developing and developed countries, supporting this finding. Steckel et al. (2021) report an inverted U-shaped incidence with middle-income groups having the largest burden in some countries of developing Asia. A contributing factor to fuel taxes disproportionately affecting richer households is that car ownership in poorer countries is limited to better-off households, thereby poor households are relatively less burdened by a fuel tax than rich ones. The review in Wang et al. (2016) and the meta-analysis in Ohlendorf et al. (2021) find that pricing instruments in the transport sector tend to be more progressive than general carbon pricing. However, results are country-specific: Steckel et al. (2021) consider 8 countries in developing Asia and find that taxing liquid fuels would be regressive in Bangladesh. The transportation mode also plays an important role when considering effects of transport subsidies. Serebrisky et al. (2009) find that public transport subsidies fail to disproportionately benefit the poor in many developing countries because many low-income households do not have access to public transport and rely on walking. Geographic characteristics are furthermore important for local policies. If low-income households live in the outskirts, they particularly benefit from public transport subsidies (Hörcher and Tirachini, 2021). Beyond consumption of different transport modes, Parry (2009) highlights the role of consuming leisure for distributional analyses in the transport sector, classifying congestion charges as regressive, particularly when reduced opportunities costs from saving travel time for higher income households are considered.

Pricing carbon emissions encompasses all fossil-fuel based consumption in energy (beyond transport, e.g. heating, electricity), and may even include other greenhouse gases such as methane. Carbon taxation tends to be regressive in developed countries, based on a recent literature review (Wang et al., 2016). Domestic energy goods such as heating and electricity tend to have income elasticities of demand below 1 in higher-income countries, contributing to regressivity on top of often progressive distributive effects from transport emissions (Pizer and Sexton, 2019). Recent literature has identified channels through which a carbon tax becomes more progressive. First, a carbon tax affects indirect emissions in household consumption and raises the prices of goods and services that use energy as an intermediate input, which often show a less regressive consumption pattern. Second, a carbon tax is less regressive when using expenditure instead of annual income as the welfare measure. Cronin et al. (2019) for the US and Feindt et al. (2021) for 23 member countries of the EU find that a comprehensive carbon tax on all emissions embodied in production is neutral to progressive when expenditure is the welfare measure. The literature review in Wang et al. (2016) and the meta-analysis in Ohlendorf et al. (2021) report a tendency of carbon pricing to be more progressive in developing countries. However, they highlight that the incidence remains country-specific, as also reported for 8 countries in developing Asia by Steckel et al. (2021).

Beyond energy and carbon taxes, a few papers show the use-side incidence of taxing other air pollutants, finding the incidence to be regressive: Johne et al. (2023) on nitrogen emissions in Germany, Parry (2004) on NO_X and SO_2 emission in the US, García-Muros et al. (2017) on local air pollutants (NO_X, SO_2 , PM10, Non-methane volatile organic compound, ammonia) in Spain, and Mardones and Mena (2020) for NO_X, SO_2 and PM emissions in Chile. More research is needed to gather systematic evidence about how the costs of air pollution policies affects inequality.

It is important to note that the incidence of other policies than pricing pollution can be quite different, as we highlight using recent research. Davis and Knittel (2019), for instance, consider fuel economy standards under the US CAFE regulation. Under this regulation, producers have to comply with a certain carbon emission footprint of their sales fleet. Using a simple model of car producer's choices combined with numerical application, they show that because standards are tradable, producers price fuel efficient cars lower than in the absence of the regulation. Fuel inefficient cars are priced higher. Because new vehicles compete with used vehicles, price changes in the two markets are linked. Davis and Knittel find that fuel economy standards have a mildly regressive effect because prices of used vehicles increase in response to the regulation, which disproportionately impacts low income households. Levinson (2019) studies energy efficiency standards in a simple theoretical model with numerical evaluation based on data from the US CAFE regulations. The author argues that an energy efficiency standard is weakly more regressive than an energy tax. As the basic argument, Levinson shows that an energy efficiency standard is like a tax on inefficient appliances. If both the inefficiency tax and the energy tax raise the same revenue, high-income households pay less under the inefficiency tax compared to the energy tax because they have more energy efficient appliances. Zhao and Mattauch (2022) extend the model of Levinson by recognizing that appliances with the same energy service (such as miles travelled per gallon) may differ in the quality of the service (such as driving a mile in an SUV versus on a moped). They show that standards can be less regressive than taxes if high-income households have preferences for more polluting service qualities.

4.2 Source-side effects

The part of policy costs that are borne by producers affects households through changes in factor incomes. The total source-side effect is denoted with ΔY_i^{P} in Eq. (2b). To understand the distributive effects on the source-side, consider a pollution tax and inelastic supply of labor and capital by households. Rausch and Schwarz (2016) show that the welfare change from source-side effects relative to income is $\Delta Y_i^{\mathrm{P}}/Y_i = \frac{\Delta w}{w} S_i^L + \frac{\Delta r}{r} S_i^K$. The relative change of the wage rate $\Delta w/w$ and of the capital rental rate $\Delta r/r$ affect income from both sources through each household's share of the factor in total income: $S_i^L = wL_i/Y_i$ for labor and $S_i^K = rK_i/Y_i$ for capital.

According to Fullerton and Metcalf (2002) and Fullerton and Heutel (2007) the distributional impact on the source-side side depends on whose relative factor price declines compared to the other. We call the factor that bears the larger burden from the policy the pollution-exposed factor. Fullerton and Heutel (2007) show that if the polluting sector is relatively more capital intensive or labor is a better substitute for pollution (or both), the relative capital rental rate tends to decline compared to the relative wage rate and capital tends to be the pollution-exposed factor.

To summarize the source-side effect, one can adopt an income elasticity approach similar to that for use-side effects. To do so, we introduce the *income elasticity of income from the pollution-exposed factor*. This elasticity measures how much income from the factor that is exposed to the policy increases when household income increases: $\eta^F = \mathrm{d}\ln(p_F F_i)/\mathrm{d}\ln(Y_i)$, where p_F is the price of the pollution exposed factor (either wage or capital rental rate) and F_i is the endowment with the pollution exposed factor (either labor or capital). If $\eta_F > 1$, source-side effects tend to be progressive.[9] In this case, households with high income rely more heavily on income from the factor that is more affected by the environmental policy, so that source-side effects have a progressive component. Likewise, source-side effects tend to be regressive if $\eta_F < 1$.

We now present evidence when environmental policy tends to have progressive or regressive effects on the source-side. As for the use-side, results strongly depend on the pollutant addressed, the type of policy in place as well as socio-economic circumstances of the specific region considered. We cannot draw general conclusions yet as the evidence is scarce and largely limited to high-income countries.

Including source-side effects within general equilibrium models often renders a carbon tax more progressive. Multiple reasons have been identified for this finding. A key driver is a larger reduction in relative capital rental rates compared to wages, at least in studies of Austria, Canada, Indonesia and the US (Rausch et al., 2011; Dissou and Siddiqui, 2014; Yusuf and Resosudarmo, 2015; Mayer et al., 2021). Rich households face a higher share of policy costs as they receive larger income shares from capital. However results are region-specific. Beck et al. (2015) find that a larger burden of a carbon tax in British Columbia falls on wages. Source-side effects are progressive because richer income deciles have larger labor income shares, given that transfer income shares decline with income. Thus, a progressive source-side effect can also be due to larger labor income shares in richer income deciles. Additionally, price changes may shift governmental transfers that are indexed by inflation, which has a progressive effect in the

[9] To illustrate this, we follow Fullerton and Heutel (2007) and Rausch and Schwarz (2016) and split total income as the sum of labor, capital and transfer income: $Y_i = wL_i + rK_i + T_i$ (factors are supplied inelastically by households). For the sake of argument, assume that capital is the pollution-exposed factor so that $\frac{\Delta r}{r} - \frac{\Delta w}{w} < 0$. The income elasticity of income from capital is $\eta^F = \mathrm{d}\ln(rK_i)/\mathrm{d}\ln(Y_i)$ so that $rK_i = AY_i^{\eta_F}$. The source-side effect relative to income is: $\frac{\Delta Y_i}{Y_i} = \frac{\Delta w}{w}\left(1 - \frac{T_i}{Y_i}\right) + \left(\frac{\Delta r}{r} - \frac{\Delta w}{w}\right) AY_i^{\eta_F - 1}$. If $\eta_F > 1$, the second part of the relative burden declines in income so that the source-side effects tend to be progressive (source-side effects would be progressive if transfer income were proportional to income so that $\frac{T_i}{Y_i}$ is independent of income). As can be seen, source-side effects could have a more nuanced shape depending on the absolute values of the elasticity, income shares and price changes.

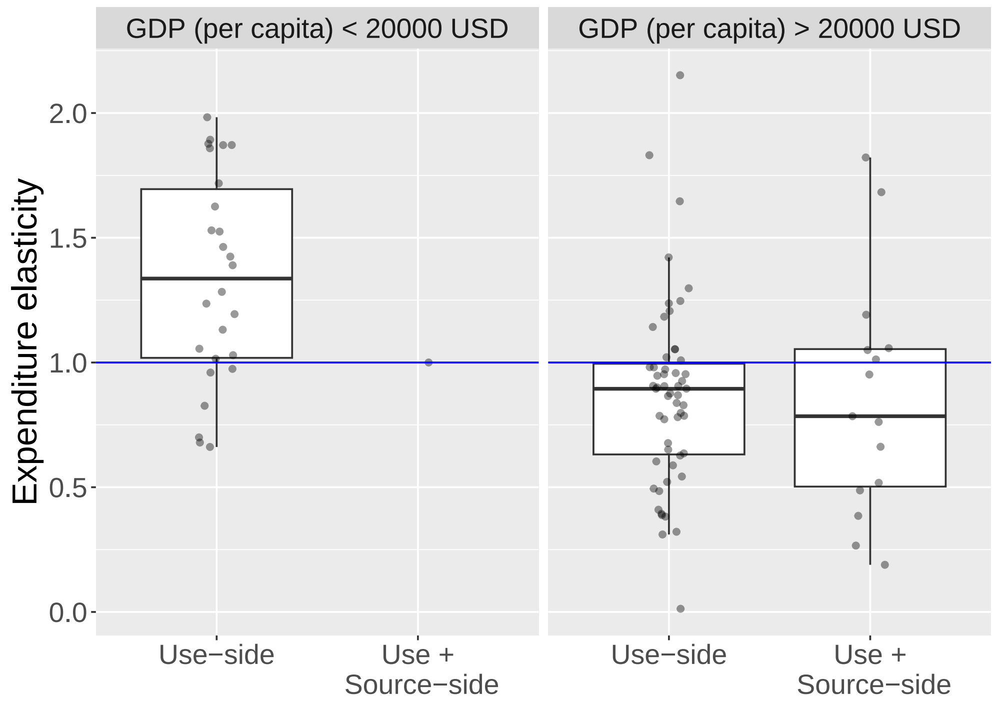

Figure 2: Expenditure elasticities of policy costs from carbon and fuel taxation (before revenue recycling) by Budolfson et al. (2021), based on 63 original incidence studies. Estimates are distinguished by national income and by modeling choice of the underlying literature (only 1 estimate in lower income category that includes source-side effects). Estimates below (above) the unity-line imply that policy costs are distributed pro-rich (pro-poor).

US (Rausch et al., 2010; Goulder et al., 2019; Cronin et al., 2019) that is separate from revenue recycling discussed in Section 5.1.

Fullerton and Monti (2013) further differentiate between high- and low-skilled workers in a theoretical model (but exclude capital as a production factor). They show that wages of high-skilled workers increase relative to low skilled workers when dirty production relies more on low-skilled workers or when high-skilled work is a better substitute for pollution. A numerical illustration with US data shows that the gross wage of low-skilled workers decreases relative to high-skilled workers in the majority of considered scenarios, which contributes to regressivity of a carbon tax. Yusuf and Resosudarmo (2015) further differentiate factor incomes in a CGE analysis of a carbon tax. They show that wages of unskilled workers particularly in the agricultural sector are reduced relatively less than wages of skilled workers, which contributes to the progressivity of a carbon tax in Indonesia. In addition, returns to capital fall relative to returns to land, which also adds to the tax's progressivity.

Budolfson et al. (2021) provide a summary of climate policy incidence by reviewing the cost incidence literature of fuel and carbon taxation. The authors extract 97 policy cost distributions among households from 63 studies (before tax revenue recycling) and compute the corresponding expenditure elasticities by matching costs distributions to consumption distributions by study country. The included studies either only calculate the use-side incidence or combine use-side and source-side effects in one expenditure elasticity of policy costs. Figure 2 shows the elasticity levels. Each data point represents the expenditure elasticity of policy costs for a specific country at a specific year. Estimates range between 0 and 2. Budolfson et al. (2021) confirm the previous finding that climate policy costs tend to be more progressive in lower income countries: Figure 2 shows elasticity estimates above 1 for the majority of studies on countries with average per capita GDP below 20000 USD, while estimates are mostly below 1 in countries with higher income. From the data, there is no clear trend that including source-side effect renders climate policy more or less regressive, however the evidence is limited.

Apart from carbon taxes, Chen et al. (2022) study the distributional effect of raising taxes on SO_2 and NOx emissions in China. Taking account of labor and capital income in a general equilibrium model, they show that low-income groups suffer a larger relative income loss than high-income groups due to the policy change.

Source-side effects can also look quite different for other, non-tax instruments. Fullerton and Heutel (2010) show that pollution intensity standards act like a pollution tax combined with an implicit output subsidy, benefiting pollution-exposed factors relative to a tax-only scenario. Under cap-and-trade systems with grandfathered permits and pollution quotas scarcity rents accrue to whoever is the owner of the restricted quantity. Parry (2004) shows that grandfathered permits in US cap-and-trade systems for SO_2, NOX, and CO_2 emissions disproportionately benefit high-income households that own more capital in affected industries.

While much of the literature on source-side effects relies on general equilibrium modeling, some recent papers estimate distributional effects of environmental regulation using ex post data of income inequality. Here, empirical estimates of income will combine policy costs on income with market-mediated effects of environmental quality on income, as discussed in Section 3.2. Huang and Yao (2023) study an SO_2 regulation implemented in China in 1998. Based on a difference-in-difference approach and a panel dataset, the study finds that stricter regulation decreased income inequality. The effect is driven by an income decline of high-income households while low-income households experienced no detectable change. Jha et al. (2019) investigate particulate matter and ozone pollution control under the US Clean Air Act. In a statistical panel from the period 2005–2015, a difference-in-difference approach shows the policy's impact on income inequality, where counties that attained air pollution standards are matched to counties that did not and faced stricter environmental regulation. Non-attainment counties experienced a substantial increase in inequality as measured by the Gini coefficient.

Another strand of literature focuses on the distributional effects of environmental policy among workers using ex-post data. Focusing only on the labor market, studies in this area do not report the effect of an environmental regulation on inequality but report relevant indirect effects. Walker (2013) studies the 1990 Clean Air Act Amendments. In a difference-in-difference-in-difference approach, Walker takes advantage of the fact that only some counties newly moved to non-attainment of PM10 and ozone pollution standards due to the 1990 regulatory changes. The study shows that workers with higher earnings experienced a larger loss in earnings than lower-income workers in the years after 1990. Yip (2018) studies a carbon tax in British Columbia using a difference-in-difference approach with other Canadian provinces. Yip shows that highly educated workers suffer less from unemployment spells after the tax than workers with lower education. Krause (2024) estimates the effect of a 1% job loss in the US Appalaichian coal industry using a long-difference approach. Men with only a high-school education experienced a decline in wages while there was no detectable wage effect for other workers. Curtis and Marinescu (2023) study millions of online job postings in the US. The authors find that new jobs in the green industry (solar and wind) have a wage premium of 21% compared to all job postings and that this wage premium is (i) higher than in the fossil fuel industry (ii) higher for jobs that require a low level of education.

5 Redistributive effects

The above sections discuss how the costs and benefits of environmental policy are distributed. We now consider how income redistribution interacts with environmental policy. Two aspects of this relationship have been the focus of the economics literature, which we discuss in turn: First, we consider the distributional effects of compensatory instruments which may be specifically designed to accompany environmental policy measures. Second, we consider the potential effects of standalone income redistribution on environmental outcomes.

5.1 Complementing environmental policy by redistribution

Environmental policy can be combined with compensatory measures, often aimed at mitigating the policy costs discussed in Section 4. These measures alter the final distribution of (net) costs and benefits and may contribute to the public acceptance of the combined policy package (Mildenberger et al., 2022).

One important class of compensatory measures are those that recycle revenues collected by environmental pricing schemes. This revenue recycling can take various forms, such as direct cash payments which alter incomes (represented by Δy_i^R in Eq. (2c)), subsidies which alter prices ($\Delta \mathbf{p}^R$), tax cuts which alter incomes and/or prices (Δy_i and/or $\mathbf{p}^R$), or additional investments in environmental quality (ΔE). Much of the literature on compensating redistribution takes estimates of policy cost incidence (see Section 4), as a starting point to then calculate net distributional effects after compensation is added.

Consider the revenues from an environmental tax, for example on carbon emissions. Recall that the *income elasticity of dirty demand* (η^D) shapes the use-side incidence of the resulting price changes. As shown in Table 1, we have $\eta^D < 1$ for carbon-intensive goods, so that the initial use-side effects are regressive, falling disproportionately on lower income households. Adding to revenue recycling can substantially mitigate this initial regressivity and sometimes even achieve net progressivity, as shown for a collection of environmental tax reforms simulated in the United States (Metcalf, 1999). More recent work incorporates general equilibrium effects in microsimulations based on household-level expenditure and income data in order to capture both use-side and source-side costs as well as the benefits of revenue recycling. An example of this approach is Rausch et al. (2011) who pair data from the 2006 U.S. Consumer Expenditure Survey with a multi-region and multi-sector general equilibrium model for the U.S. economy. The authors find that, while uniform lump-sum rebates of carbon tax revenue, sometimes called 'carbon dividends', are strongly progressive, using the revenue to finance income or capital tax cuts is relatively more regressive. Importantly, the lump-sum payments can result in (net) progressivity despite the initial regressivity of use-side costs. The intuition is simple: While lower income households spend larger shares of incomes on carbon-intensive goods ($\eta^D < 1$), they tend to spend smaller total amounts on those goods ($0 < \eta^D$), thus also contributing less to the carbon tax revenue while receiving the same average lump-sum payment.

This progressivity of carbon dividends has more recently been confirmed at the multinational and global level. For example, Sager (2023) calibrates a global non-homothetic demand system using a trade gravity approach, paired with input-output based emission accounting to simulate welfare effects of carbon pricing across the global income distribution. The results show that adding carbon dividends to a global carbon price leads to net progressive effects, both within nations and globally. Similarly, Feindt et al. (2021) find progressive effects of per-capita dividends in a microsimulation based on household expenditure surveys in 23 EU member states. Of course, transfer payments could be targeted to specifically benefit the poor or other disadvantaged groups. With reference to Latin America and the Caribbean, Vogt-Schilb et al. (2019) demonstrate that only 30% of carbon tax revenue needs to be returned to the poor to offset the adverse effects of carbon taxation.

Revenue recycling can also be used to lower taxes or reduce prices. In contrast to lump-sum transfers, pairing a carbon tax with income tax reductions may be insufficient to compensate low-income households who have lower relative income tax burdens, as shown theoretically by (Fullerton and Monti, 2013) in line with the microsimulation results by Rausch et al. (2011). However, ex post empirical evidence suggests that targeted income tax reductions did render British Columbia's carbon tax system moderately progressive (Murray and Rivers, 2015). In another example from Germany, lowering taxes on electricity, a necessity good, was shown to be progressive (Neuhoff et al., 2013).

A relatively new insight in this literature is that (progressive) revenue recycling can generate additional benefits in suboptimal tax systems (Klenert et al., 2018; Budolfson et al., 2021). Jacobs and De Mooij (2015) extend standard models of optimal taxation in the tradition of Mirrlees (1971) to incorporate environmental damages and heterogeneous agents. They show that, when the tax system is optimized to satisfy redistributive objectives, the second-best corrective tax on an environmental externality can equal marginal damages and no longer needs to be adjusted for the marginal cost of public funds, as was the case in models with a representative agent (Sandmo, 1975; Bovenberg and van der Ploeg, 1994). Ultimately, optimal revenue use will depend on the pre-existing tax system and welfare objectives, possibly giving rise to 'hybrid' solutions. For example, Fried et al. (2021) propose a combination of capital income tax and labor income tax reductions to both alleviate distortions and achieve redistributive goals.

As in previous sections, we can conceptualize the incidence of revenue-recycling measures using specific income elasticities. The above discussion suggests an important role for an *income elasticity of income from subsidized factors or transfers* (η^T) when revenues are recycled through income transfers or tax reductions (e.g. on individual income). If $\eta^T < 1$, a transfer will tend to represent a higher relative share in the income of lower income households, and thus be progressive. A uniform lump-sum payment such as a carbon dividend is characterized by $\eta^T = 0$ and thus highly progressive. Meanwhile, using revenues to cut income taxes, which tend to be disproportionately paid by higher income households ($\eta^T > 1$), is relatively more regressive. Similarly, if revenue recycling takes the form of subsidizing certain goods or cutting consumption taxes, we would consider an *income elasticity of subsidized demand* (η^{C_S}). And if revenues are used to finance public goods, such as public infrastructure, the benefit distribution will depend on an *income elasticity of WTP for public goods* (η^P).

5.2 Environmental effects of income redistribution

Even income redistribution that is not linked to environmental policy can have important repercussions for the environment. For example, it is a common hypothesis that higher levels of economic inequality are themselves a cause of more environmental degradation or less stringent environmental policy (e.g., Stiglitz, 2012). However, it is unclear—from both economic theory and the empirical evidence—whether there is indeed a systematic relationship. To provide a conceptual footing for this still relatively under-explored area, we distinguish three channels through which income inequality may shape environmental outcomes: (1) consumer demand, (2) collective action and public good provision, and (3) political power.

The consumer demand channel (1) is a story of aggregation over non-linear expenditures. Consumers at different income levels demand bundles of goods with varying environmental intensities. The relationship between household income and average environmental footprints can be represented by Environmental Engel Curves (EEC). Levinson and O'Brien (2019) construct EEC's for local air pollutants, such as particulate matter, by matching household-level consumption data from the U.S. Consumer Expenditure Survey with emission factors for industries calculated using input-output accounting methods. They then plot the relationship between after-tax income and average pollution embedded in household consumption, at times controlling for other household characteristics. They find EEC's for air pollutants that are upward-sloping and concave (as in the left panel of Figure 3).

Concave EEC's suggest an *income elasticity of dirty demand* (η^D) below unity and non-linear EEC's give rise to an aggregation property where the distribution of income

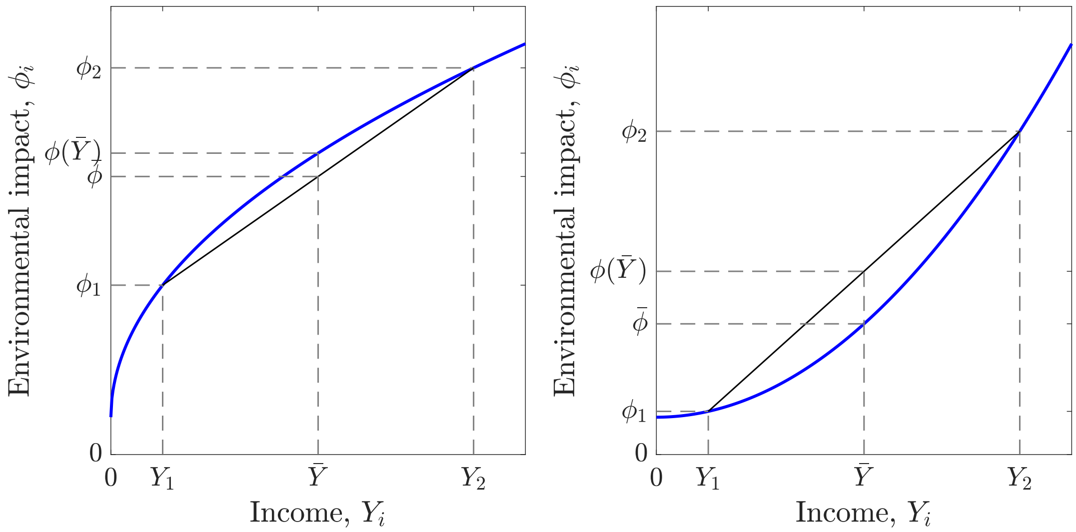

Figure 3: Environmental Engel curves and the "equity-pollution dilemma" (Sager, 2019). Under concave Environmental Engel curves (left), income dispersion (Y_1, Y_2) leads to lower environmental impact (ϕ) than equality ($\overline{Y}$).

shapes aggregate environmental outcomes (Scruggs, 1998; Heerink et al., 2001). This is demonstrated empirically by Sager (2019), who applies the methodology of Levinson and O'Brien (2019) to greenhouse gas emissions and finds upward-sloping and concave EEC's for greenhouse gas emissions embedded in the consumption of U.S. households. Consider a progressive income transfer from a richer (Y_2) to a poorer household (Y_1). The propensity to generate emissions by spending additional income on polluting goods (ϕ), shown by the slope of the EEC, is higher at lower income levels. This gives rise to an "equity-pollution dilemma" as formulated by Sager (2019): Progressive redistribution raises demand for the polluting good.[10] In the extreme, complete income inequality where both households have income $\overline{Y}$ gives rise to the highest possible per capita environmental footprint, $\phi(\overline{Y})$. As already discussed in Table 1, *income elasticities of dirty demand* are often below 1, suggesting that concave EEC's are common.

Inequality can also shape environmental outcomes via collective action dynamics (2). For example, it can alter society's valuation of environmental public goods through a similar aggregation property: If the *income elasticity of WTP for environmental quality* (η^W) is below unity, as seems common in many settings (see Figure 1), more equal societies will exhibit a higher aggregate WTP for environmental public goods, all else equal (Baumgärtner et al., 2017; Drupp et al., 2018; Meya et al., 2020). Again for $(\eta^W) > 1$ the effect would be reverse and aggregate WTP would decrease with income equality.

[10] To show this formally, assume demand for a single polluting good D is a function of price and income, $D_i = f(p_D) \cdot Y_i^{\eta^D}$. A small Pigou-Dalton income transfer of x from a rich household (Y_2) to a poorer one (Y_1) alters aggregate demand for D by approximately $f(p_D) \cdot x\eta^D \left[Y_1^{(\eta^D-1)} - Y_2^{(\eta^D-1)} \right]$, which is positive when $0 < \eta^D < 1$ and $f(p_D) \geq 0$.

But even holding aggregate WTP constant, the distribution of resource endowments may alter strategic incentives to contribute towards environmental public goods and the sustainable management of common-pool resources such as fisheries or forests. The underlying logic is set forth by Olson (1965) who argues that in large groups that are more equal, the benefits of public good provision are diffuse and members receive similar, relatively small benefits. Meanwhile, the costs of engaging in and organizing collective action are relatively high. By contrast, per-person benefits are higher in small groups and in groups with unequal access to the (impure) public good.

The implications of inequality for a group's ability to manage a common-pool resource, such as a fishery, are more ambiguous (Baland and Platteau, 1999). Dayton-Johnson and Bardhan (2002) show that in a non-cooperative common-pool resource game, the relationship between inequality and preservation levels is U-shaped. Both very low and very high levels of inequality can in theory favor high levels of preservation, either because many fishermen have sufficiently large stakes, or because a few dominate. And inequality may erode trust and social capital that have been shown to underpin successful common-pool resource management (Ostrom, 2009). The experimental evidence from public good games tends to find that inequality does lower group cooperation (Anderson et al., 2008; Tavoni et al., 2011; Gächter et al., 2017), which would translate into less ambitious environmental policy all else equal.

Finally, the political power channel (3) associates the distribution of economic means with the distribution of political influence (Torras and Boyce, 1998). For example, if richer citizens have more political influence and weaker preferences for environmental public goods, more inequality would arguably result in less political demand for environmental policy. The political power channel has not received much attention from economists to date, but is a topic of research in environmental studies and related disciplines (summarized in Cushing et al., 2015). To our knowledge, there is no clear empirical evidence for the claim that high-income citizens have less concern for the environment, although this could be measured using the income elasticity of WTP, η^W (see Section 3). Meanwhile, there is ample evidence for the influence of political processes on the provision of local public goods and that low-income and otherwise disadvantaged communities are more likely to be exposed to toxic pollution (e.g., Hamilton, 1995; Brooks and Sethi, 2017) while receiving lower levels of compensation (Timmins and Vissing, 2022). The limited empirical evidence that does exist regarding the political power channel comes from cross-sectional and aggregate-level analyses, such as Boyce et al. (1999) who find an association between higher levels of power inequality and weaker environmental policy stringency across U.S. states.

Besides specific mechanisms, there is a literature that looks for correlations between income inequality and environmental degradation at aggregate levels, as surveyed by Berthe and Elie (2015). For example, Baek and Gweisah (2013) find a positive association between Gini index values and per capita CO2 emissions across U.S. states. Similarly, there is some evidence of a positive correlation between measures of within-country inequality and local air pollution levels (Torras and Boyce, 1998) as well as biodiversity loss

(Mikkelson et al., 2007). Those findings seem to support the negative environmental effects of inequality hypothesized by proponents of the political power or collective action channels. On the other hand, multiple studies have found negative correlations between country-level income inequality and per capita carbon emissions (Ravallion et al., 2000; Heerink et al., 2001; Coondoo and Dinda, 2008), more in line with the consumption-based "equity-pollution dilemma". The conflicting results are indicative of a key problem with empirical investigations of the inequality-environment relationship, which are difficult to interpret causally due to the many factors which co-vary with both inequality and environmental outcomes.

At the heart of these three channels through which inequality relates to environmental outcomes, we can again identify an important role for income elasticities. We are most concerned with regressive effects when policy targets necessity goods so that the *income elasticity of dirty demand* (η^D) lies below 1. But it is also in those same situations that we are more likely to face an "equity-pollution dilemma" when adding progressive compensation, a kind of distributional rebound effect. Similarly, the public good channel is shaped by preference aggregation over the *income elasticity of WTP for environmental public goods* (η^W) although this is rarely formalized in the literature to date.

6 Discussion and Conclusion

Several common themes emerge from our synthesis of the literature. Conceptually, income elasticities facilitate a common framework that connects changes in environmental policy to economic inequality, in ways we summarize in Table 2. The distribution of environmental improvements, for instance, is determined largely by the income elasticity of WTP for that change, which is mediated by further income elasticities relating to exposure and adaptation (Section 3.1). If the income elasticity of WTP is larger than unity, environmental benefits are progressive, and if the elasticity is smaller than unity, benefits are regressive. The distribution of policy costs in turn is driven by the income elasticity of dirty demand (Section 4.1) and the income elasticity of income from pollution-exposed factors (Section 4.2). Policy costs tend to be regressive if these elasticities are smaller than unity, and progressive when elasticities exceed unity. The environmental effects of income redistribution, too, are shaped by the income elasticity of dirty demand. If it lies below unity, we face an "equity-pollution dilemma" (Section 5). Thus, we recommend that empirical studies more systematically report estimates of income elasticities where applicable to facilitate synthesis, allow for meaningful comparisons of effects across settings, and to provide input parameters for policy simulations.

Another insight from our synthesis concerns measurement. There is far more research on the distribution of economic resources—in terms of consumption, income and wealth—than on the distribution of environmental goods such as clean water, access to urban green spaces, or opportunities for recreation in biodiverse landscapes (Section 3.1). Economic resources are often easier to observe and measure. Yet, recent advances in measuring environmental quality with improved precision and granular-

Table 2: Income Elasticities and Distributional Effects

	INCOME ELASTICITY OF Demand for	Income from	WTP for
Direct env. benefits (Sec. 3.1)			Env. Quality $\eta^W < 1$: progressive $\eta^W > 1$: regressive
Market-mediated env. benefits (Sec. 3.2)	Env. exposed goods $\eta^{C_E} < 1$: regressive $\eta^{C_E} > 1$: progressive	Env. exposed factors $\eta^{F_E} < 1$: progressive $\eta^{F_E} > 1$: regressive	
Policy costs (Sec. 4)	Dirty goods $\eta^D < 1$: regressive $\eta^D > 1$: progressive	Pol. exposed factors $\eta^F < 1$: regressive[a] $\eta^F > 1$: progressive[a]	
Benefits of redistribution (Sec. 5.1)	Subsidized goods $\eta^{C_S} < 1$: progressive $\eta^{C_S} > 1$: regressive	Subs. factors & transfers $\eta^T < 1$: progressive $\eta^T > 1$: regressive	Public Goods $\eta^P < 1$: progressive $\eta^P > 1$: regressive
Env. effects of redistribution (Sec. 5.2)	Dirty Goods $\eta^D < 1$: pollution $\uparrow$ $\eta^D > 1$: pollution $\downarrow$		Env. Quality $\eta^W < 1$: $\overline{WTP} \uparrow$ $\eta^W > 1$: $\overline{WTP} \downarrow$

Notes: Summary of the role of various income elasticities for distributional analysis of environmental policy. Progressive [regressive] refers to pro-poor [pro-rich] costs and benefits, i.e. benefits that fall disproportionately on those with lower [higher] incomes and cost that fall disproportionately on those with higher [lower] incomes. Details in the text.
[a] This assumes that transfer income is proportional to income (or that the factor not exposed to the policy is the numeraire), see Sec. 4.2.

ity, combined with more systematic surveys on preferences for the environment, should help close this gap. Many research opportunities remain, as the dynamics discussed in this review may vary across spatial scales and environmental domains.

In some areas, there is sufficiently strong evidence to draw conclusions about how environmental policy interacts with economic inequality. Consider the incidence of climate policy. Evidence suggests that the income elasticity of dirty demand is smaller than unity in developed countries (Section 4.1). This drives the regressive use-side incidence of carbon pricing. And because the income elasticity is almost certainly positive, recycling revenues via lump-sum carbon dividends is likely progressive (Section 5.1).

In other areas, considerable knowledge gaps remain. While a number of empirical studies investigate source-side effects of climate policy via labor markets, little is known about source-side effects via capital (Section 4.2). Furthermore, fundamental parameters are often context-dependent and the income elasticity of dirty demand appears larger in developing countries for example. This relates to an important gap identified in our review: Most empirical studies focus on the United States and, to a lesser degree, Europe. The extent to which these can be extrapolated to other countries, in particular developing ones, remains to be investigated. Similarly, not all environmental domains are equally well-studied. Our review has largely focused on climate, energy and air pollution examples. This choice was dictated by availability: More research is needed on benefit and cost incidence relating to other environmental domains, such as for water quality, biodiversity, or other ecosystem services.

Another limitation in the literature is the frequent reliance on constant income elasticities, which may obscure more nuanced relationships in some contexts. For example, a constant income elasticity cannot capture an inverted U-shaped distribution of environmental policy costs where the middle income segments bear the largest burden, or capture the effects of environmental policies that make some groups better off while at the same time making others worse off. Future research should explore non-constant elasticities across the income distribution and how elasticities change with policy-stringency, across population groups and across time as well as with the evolution of production technologies. Lastly, research needs to identify cases when using income elasticities is not consistent with their findings.

Environmental policy appraisal also depends on how pre-existing inequalities are weighted in the social welfare function (Eq. (2c)). While common, unweighted benefit-cost analysis rests on strong assumptions regarding both individual utility (i.e. quasi-linear utility with constant marginal utility of income) and the social welfare function (i.e. a utilitarian welfare function without inequality aversion). One explanation for the infrequent use of equity weights is the lack of ready-to-use parameters (Fleurbaey and Abi-Rafeh, 2016), suggesting a need for more empirical studies that elicit both individual and social preferences. Such work is emerging in some areas, for example in climate economics that estimates the social cost of carbon, although the use of distributional weights is usually restricted to income inequality between regions, ignoring inequality within regions. Del Campo et al. (2024) review 24 studies that estimate social inequality aversion and find a positive degree of inequality-aversion ($\rho > 0$) in the common setting of a constant relative inequality-aversion social welfare function (Section 2).

Beyond equity weights that take into account pre-existing economic inequalities, it would be useful to investigate environmental distributional weights that take into account how marginal utility of income varies with the pre-existing endowment of environmental goods (Adler, 2016; Meya, 2020). These are likely more important for environmental goods that are essential to human well-being, where supplies do not rise above subsistence levels or levels at which people are reluctant to substitute them by market-traded consumption goods. Central parameters are thus not only the elasticity of marginal utility with respect to consumption (often referred to as the "intratemporal consumption inequality aversion", e.g. Groom and Maddison (2019)) but also the elasticity of the marginal utility of income with respect to environmental goods. Venmans and Groom (2021) present a first experimental study to elicit environmental inequality aversion. Beyond refining standard welfare analysis, alternative normative frameworks—besides the workhorse anthropocentric, utilitarian approach—may place more emphasis on animal welfare and the "intrinsic value" of nature (Carlier and Treich, 2020).

Our review also highlights the importance of interdependencies and feedback effects. While there is a lot of research on each inequality-environment linkage that we consider, few studies engage with multiple and none consider all. Such omissions can generate flawed policy analysis. Consider again the case of carbon pricing. An analysis of the distributional effects of a carbon tax (Section 4) raises the issue of regressive consumer costs driven by the income elasticity of dirty demand. It also points to revenue recycling and income effects as additional elements through which greater progressivity can be achieved (Section 5.1). Here, the literature has focused on the net effect of use-side (and rarely also source-side) effects due to carbon pricing along with the effect of transfer payments, while the distribution of direct and market-mediated environmental benefits is often not explicitly taken into account. Furthermore, shifting income towards low-income households, who have a higher propensity to spend it on carbon-intensive goods, may counteract the emissions reductions from pricing carbon (Section 5.2). Accounting for such interlinkages will likely alter the optimal design of both instruments.

Another case in point is the issue of poor air quality in low-income neighborhoods (Section 3). We may interpret this as a mere manifestation of income inequality, with the result of lower-income households investing less in local air quality. Policy makers could then focus on income redistribution alone. But lower air quality may lead to higher healthcare expenditures or lower levels of human capital accumulation (e.g., in terms of the ability to learn and thus education) amongst low-income households. Such externalities would exacerbate the effects of the unequal distribution of air quality. Moreover, low-income households may find it more difficult to influence the political process, which might reinforce a vicious cycle (Section 5.2). Explicit consideration of the co-variation of income and pollution may alter benefit-cost analyses of measures to improve and/or to spatially re-allocate air quality.

In sum, while we have a solid theoretical and oftentimes empirical understanding of selected key links between environmental change and economic inequality, many gaps remain. First, research should focus on further investigating individual inequality-environment links. This includes empirically capturing the novel income elasticities we identify and summarize in Table 2 as well as assessing inequality-environment links across different temporal and spatial scales. Second, the normative assumptions underlying economic welfare analysis should be made explicit and scrutinized. Third, more researh is required to investigate distributional effects of environmental policies beyond climate and air pollution and in developing countries. Finally, economic research will have to better integrate analyses of multiple inequality-environment interlinkages to better understand the full distributional effects of environmental policies or the environmental effects of redistribution. Filling these gaps is a prerequisite for economists to inform policy-makers who increasingly require such integrated analyses when trying to implement key public policy aims, such as the UN Sustainable Development Goals.

References

Adler, M. D. (2016). Benefit–cost analysis and distributional weights: An overview. *Review of Environmental Economics and Policy*.

Aguilar-Gomez, S., H. Dwyer, J. Graff Zivin, and M. Neidell (2022). This is air: The "nonhealth" effects of air pollution. *Annual Review of Resource Economics 14*, 403–425.

Anand, S. and P. Segal (2008). What do we know about global income inequality? *Journal of Economic Literature 46*(1), 57–94.

Anderson, L. R., J. M. Mellor, and J. Milyo (2008). Inequality and public good provision: An experimental analysis. *The Journal of Socio-Economics 37*(3), 1010–1028.

Andersson, J. J. (2019). Carbon taxes and co 2 emissions: Sweden as a case study. *American Economic Journal: Economic Policy 11*(4), 1–30.

Atkinson, A. B. (1970). On the measurement of inequality. *Journal of economic theory 2*(3), 244–263.

Baek, J. and G. Gweisah (2013). Does income inequality harm the environment?: Empirical evidence from the united states. *Energy Policy 62*, 1434–1437.

Baland, J.-M. and J.-P. Platteau (1999). The ambiguous impact of inequality on local resource management. *World development 27*(5), 773–788.

Banzhaf, H. S. and R. P. Walsh (2008). Do people vote with their feet? an empirical test of tiebout's mechanism. *American Economic Review 98*(3), 843–863.

Banzhaf, S., L. Ma, and C. Timmins (2019). Environmental justice: The economics of race, place, and pollution. *Journal of Economic Perspectives 33*(1), 185–208.

Barbier, E. B., M. Czajkowski, and N. Hanley (2017). Is the income elasticity of the willingness to pay for pollution control constant? *Environmental and Resource Economics 68*, 663–682.

Basaglia, P., S. M. Behr, and M. A. Drupp (2024). De-fueling externalities: Causal effects of fuel taxation and mediating mechanisms for reducing climate and pollution costs. Technical report, CESifo Working Paper No. 10508.

Baumgärtner, S., M. A. Drupp, J. N. Meya, J. M. Munz, and M. F. Quaas (2017). Income inequality and willingness to pay for environmental public goods. *Journal of Environmental Economics and Management 85*, 35–61.

Beck, M., N. Rivers, R. Wigle, and H. Yonezawa (2015). Carbon tax and revenue recycling: Impacts on households in british columbia. *Resource and Energy Economics 41*, 40–69.

Bento, A., M. Freedman, and C. Lang (2015). Who benefits from environmental regulation? evidence from the clean air act amendments. *Review of Economics and Statistics 97*(3), 610–622.

Bento, A. M. (2013). Equity impacts of environmental policy. *Annu. Rev. Resour. Econ. 5*(1), 181–196.

Berthe, A. and L. Elie (2015). Mechanisms explaining the impact of economic inequality on environmental deterioration. *Ecological economics 116*, 191–200.

Bovenberg, A. L. and F. van der Ploeg (1994). Environmental policy, public finance and the labour market in a second-best world. *Journal of Public Economics 55*(3), 349–390.

Boyce, J. K., A. R. Klemer, P. H. Templet, and C. E. Willis (1999). Power distribution, the environment, and public health: A state-level analysis. *Ecological Economics 29*(1), 127–140.

Brooks, N. and R. Sethi (2017). The distribution of pollution: community characteristics and exposure to air toxics. In *Distributional Effects of Environmental and Energy Policy*, pp. 301–318. Routledge.

Brueckner, J. K., J.-F. Thisse, and Y. Zenou (1999). Why is central paris rich and downtown detroit poor?: An amenity-based theory. *European Economic Review 43*(1), 91–107.

Budolfson, M., F. Dennig, F. Errickson, S. Feindt, M. Ferranna, M. Fleurbaey, D. Klenert, U. Kornek, K. Kuruc, A. Méjean, et al. (2021). Climate action with revenue recycling has benefits for poverty, inequality and well-being. *Nature Climate Change 11*(12), 1111–1116.

Burke, M., S. M. Hsiang, and E. Miguel (2015). Global non-linear effect of temperature on economic production. *Nature 527*(7577), 235–239.

Cain, L., D. Hernandez-Cortes, C. Timmins, and P. Weber (2023). Recent findings and methodologies in economics research in environmental justice. *Review of Environmental Economics and Policy*.

Campa, P. and L. Muehlenbachs (2023). Addressing environmental justice through in-kind court settlements. *American Economic Journal: Economic Policy*, cond. accepted.

Carleton, T., A. Jina, M. Delgado, M. Greenstone, T. Houser, S. Hsiang, A. Hultgren, R. E. Kopp, K. E. McCusker, I. Nath, et al. (2022). Valuing the global mortality consequences of climate change accounting for adaptation costs and benefits. *The Quarterly Journal of Economics 137*(4), 2037–2105.

Carlier, A. and N. Treich (2020). Directly valuing animal welfare in (environmental) economics. *International Review of Environmental and Resource Economics 14*(1), 113–152.

Chen, S., Y. Chen, Z. Lei, and J.-S. Tan-Soo (2020). Impact of air pollution on short-term movements: evidence from air travels in china. *Journal of Economic Geography 20*(4), 939–968.

Chen, S., P. Oliva, and P. Zhang (2022). The effect of air pollution on migration: evidence from china. *Journal of Development Economics 156*, 102833.

Chen, Z., R. Zhang, F. Wang, F. Xia, B. Liu, and B. Zhang (2022). The distributional effects of china'senvironmental taxation: A multi-regional analysis. *Journal of Environmental Management 324*, 116276.

Christensen, P., I. Sarmiento-Barbieri, and C. Timmins (2021). Racial discrimination and housing outcomes in the united states rental market. Technical report, National Bureau of Economic Research.

Christensen, P. and C. Timmins (2022). Sorting or steering: The effects of housing discrimination on neighborhood choice. *Journal of Political Economy 130*(8), 2110–2163.

Cohen, F. and A. Dechezleprêtre (2022). Mortality, temperature, and public health provision: evidence from mexico. *American Economic Journal: Economic Policy 14*(2), 161–192.

Coondoo, D. and S. Dinda (2008). Carbon dioxide emission and income: A temporal analysis of cross-country distributional patterns. *Ecological Economics 65*(2), 375–385.

Cronin, J. A., D. Fullerton, and S. Sexton (2019). Vertical and horizontal redistributions from a carbon tax and rebate. *Journal of the Association of Environmental and Resource Economists 6*(S1), S169–S208.

Curtis, E. M. and I. Marinescu (2023). Green energy jobs in the united states: What are they, and where are they? *Environmental and Energy Policy and the Economy 4*(1), 202–237.

Cushing, L., R. Morello-Frosch, M. Wander, and M. Pastor (2015). The haves, the have-nots, and the health of everyone: the relationship between social inequality and environmental quality. *Annual Review of Public Health 36*, 193–209.

Dasgupta, P. (2021). *The economics of biodiversity: the Dasgupta review.* Hm Treasury.

Davis, L. W. and C. R. Knittel (2019). Are fuel economy standards regressive? *Journal of the Association of Environmental and Resource Economists 6*(S1), S37–S63.

Dayton-Johnson, J. and P. Bardhan (2002). Inequality and conservation on the local commons: A theoretical exercise. *The Economic Journal 112*(481), 577–602.

De Lima, C. Z., J. R. Buzan, F. C. Moore, U. L. C. Baldos, M. Huber, and T. W. Hertel (2021). Heat stress on agricultural workers exacerbates crop impacts of climate change. *Environmental Research Letters 16*(4), 044020.

Del Campo, S., D. Anthoff, and U. Kornek (2024). Inequality aversion for climate policy. *Review of Environmental Economics and Policy 18*.

Depro, B., C. Timmins, and M. O'Neil (2015). White flight and coming to the nuisance: can residential mobility explain environmental injustice? *Journal of the Association of Environmental and resource Economists 2*(3), 439–468.

Dillender, M. (2021). Climate change and occupational health: Are there limits to our ability to adapt? *Journal of Human Resources 56*(1), 184–224.

Dissou, Y. and M. S. Siddiqui (2014). Can carbon taxes be progressive? *Energy Economics 42*, 88–100.

Dorband, I. I., M. Jakob, M. Kalkuhl, and J. C. Steckel (2019). Poverty and distributional effects of carbon pricing in low-and middle-income countries–a global comparative analysis. *World Development 115*, 246–257.

Douenne, T. (2020). The vertical and horizontal distributive effects of energy taxes: A case study of a french policy. *The Energy Journal 41*(3).

Douenne, T. and A. Fabre (2022). Yellow vests, pessimistic beliefs, and carbon tax aversion. *American Economic Journal: Economic Policy 14*(1), 81–110.

Drupp, M. A. (2018). Limits to substitution between ecosystem services and manufactured goods and implications for social discounting. *Environmental and Resource Economics 69*(1), 135–158.

Drupp, M. A. and M. C. Hänsel (2021). Relative prices and climate policy: How the scarcity of nonmarket goods drives policy evaluation. *American Economic Journal: Economic Policy 13*(1), 168–201.

Drupp, M. A., J. N. Meya, S. Baumgärtner, and M. F. Quaas (2018). Economic inequality and the value of nature. *Ecological Economics 150*, 340–345.

Drupp, M. A., Z. M. Turk, B. Groom, and J. Heckenhahn (2024). Limited substitutability, relative price changes and the uplifting of public natural capital values. *arXiv preprint arXiv:2308.04400*.

Ebert, U. (2003). Environmental goods and the distribution of income. *Environmental and Resource Economics 25*(4), 435–459.

Fabra, N. and M. Reguant (2014). Pass-through of emissions costs in electricity markets. *American Economic Review 104*(9), 2872–2899.

Feindt, S., U. Kornek, J. M. Labeaga, T. Sterner, and H. Ward (2021, nov). Understanding regressivity: Challenges and opportunities of european carbon pricing. *Energy Economics 103*, 105550.

Femenia, F. et al. (2019). A meta-analysis of the price and income elasticities of food demand. *German Journal of Agricultural Economics 68*(2), 77–98.

Fleurbaey, M. (2009). Beyond gdp: The quest for a measure of social welfare. *Journal of Economic literature 47*(4), 1029–1075.

Fleurbaey, M. and R. Abi-Rafeh (2016). The use of distributional weights in benefit–cost analysis: Insights from welfare economics1. *Review of Environmental Economics and Policy*.

Flores, N. E. and R. T. Carson (1997). The relationship between the income elasticities of demand and willingness to pay. *Journal of environmental Economics and management 33*(3), 287–295.

Fried, S., K. M. Novan, and W. Peterman (2021). Recycling carbon tax revenue to maximize welfare.

Friedman, M. (1957). *Theory of the consumption function*. Princeton university press.

Frondel, M., G. Kussel, and S. Sommer (2019). Heterogeneity in the price response of residential electricity demand: A dynamic approach for germany. *Resource and Energy Economics 57*, 119–134.

Fullerton, D. (2011). Six distributional effects of environmental policy. *Risk Analysis: An International Journal 31*(6), 923–929.

Fullerton, D. and G. Heutel (2007). The general equilibrium incidence of environmental taxes. *Journal of Public Economics 91*(3-4), 571–591.

Fullerton, D. and G. Heutel (2010). The general equilibrium incidence of environmental mandates. *American Economic Journal: Economic Policy 2*(3), 64–89.

Fullerton, D. and G. Heutel (2011). Analytical general equilibrium effects of energy policy on output and factor prices. *The BE Journal of Economic Analysis & Policy 10*(2).

Fullerton, D. and G. E. Metcalf (2002). Tax incidence. *Handbook of public economics 4*, 1787–1872.

Fullerton, D. and H. Monti (2013). Can pollution tax rebates protect low-wage earners? *Journal of Environmental Economics and Management 66*(3), 539–553.

Gächter, S., F. Mengel, E. Tsakas, and A. Vostroknutov (2017). Growth and inequality in public good provision. *Journal of Public Economics 150*, 1–13.

Ganapati, S., J. S. Shapiro, and R. Walker (2020). Energy cost pass-through in us manufacturing: Estimates and implications for carbon taxes. *American Economic Journal: Applied Economics 12*(2), 303–342.

Gao, X., R. Song, and C. Timmins (2023). Information, migration, and the value of clean air. *Journal of Development Economics 163*, 103079.

García-Muros, X., M. Burguillo, M. González-Eguino, and D. Romero-Jordán (2017). Local air pollution and global climate change taxes: a distributional analysis for the case of spain. *Journal of Environmental Planning and Management 60*(3), 419–436.

Gilli, M., J. Emmerling, M. Calcaterra, and F. Granella (2024). Climate change impacts on the within-country income distributions. *Available at SSRN 4520461, http://dx.doi.org/10.2139/ssrn.4520461*.

Goulder, L. H., M. A. Hafstead, G. Kim, and X. Long (2019). Impacts of a carbon tax across us household income groups: What are the equity-efficiency trade-offs? *Journal of Public Economics 175*, 44–64.

Grainger, C. A. (2012). The distributional effects of pollution regulations: Do renters fully pay for cleaner air? *Journal of Public Economics 96*(9-10), 840–852.

Green, R., L. Cornelsen, A. D. Dangour, R. Turner, B. Shankar, M. Mazzocchi, and R. D. Smith (2013). The effect of rising food prices on food consumption: systematic review with meta-regression. *Bmj 346*.

Groom, B., M. A. Drupp, M. C. Freeman, and F. Nesje (2022). The future, now: A review of social discounting. *Annual Review of Resource Economics 14*, 467–491.

Groom, B. and D. Maddison (2019). New estimates of the elasticity of marginal utility for the uk. *Environmental and Resource Economics 72*(4), 1155–1182.

Hamilton, J. T. (1995). Testing for environmental racism: Prejudice, profits, political power? *Journal of Policy Analysis and Management 14*(1), 107–132.

Harju, J., T. Kosonen, M. Laukkanen, and K. Palanne (2022). The heterogeneous incidence of fuel carbon taxes: Evidence from station-level data. *Journal of Environmental Economics and Management 112*, 102607.

Hassett, K. A., A. Mathur, and G. E. Metcalf (2009). The incidence of a us carbon tax: A lifetime and regional analysis. *The Energy Journal 30*(2).

Hausman, C. and S. Stolper (2021). Inequality, information failures, and air pollution. *Journal of Environmental Economics and Management 110*, 102552.

Heerink, N., A. Mulatu, and E. Bulte (2001). Income inequality and the environment: aggregation bias in environmental kuznets curves. *Ecological Economics 38*(3), 359–367.

Hörcher, D. and A. Tirachini (2021). A review of public transport economics. *Economics of transportation 25*, 100196.

Hsiang, S., P. Oliva, and R. Walker (2019). The distribution of environmental damages. *Review of Environmental Economics and Policy*.

Huang, B. and Y. Yao (2023). Does environmental regulation matter for income inequality? new evidence from chinese communities. *Journal of the Association of Environmental and Resource Economists q0*(5).

Jacobs, B. and R. A. De Mooij (2015). Pigou meets mirrlees: On the irrelevance of tax distortions for the second-best pigouvian tax. *Journal of Environmental Economics and Management 71*, 90–108.

Jacobsen, J. B. and N. Hanley (2009). Are there income effects on global willingness to pay for biodiversity conservation? *Environmental and Resource Economics 43*(2), 137–160.

Jha, A., P. H. Matthews, and N. Z. Muller (2019). Does environmental policy affect income inequality? evidence from the clean air act. In *AEA Papers and Proceedings*, Volume 109, pp. 271–276. American Economic Association 2014 Broadway, Suite 305, Nashville, TN 37203.

Johansson-Stenman, O. (2005). Distributional weights in cost-benefit analysis—should we forget about them? *Land Economics 81*(3), 337–352.

Johne, C., E. Schröder, and H. Ward (2023). The distributional effects of a nitrogen tax: Evidence from germany. *Ecological Economics 208*, 107815.

Johnson, P. and C. Papageorgiou (2020). What remains of cross-country convergence? *Journal of Economic Literature 58*(1), 129–75.

Känzig, D. R. (2021). The unequal economic consequences of carbon pricing. *SSRN Working Paper*, https://ssrn.com/abstract=3786030.

Klenert, D., G. Schwerhoff, O. Edenhofer, and L. Mattauch (2018). Environmental taxation, inequality and engel's law: the double dividend of redistribution. *Environmental and Resource Economics 71*(3), 605–624.

Krause, E. (2024). Job loss, selective migration, and the accumulation of disadvantage: Evidence from appalachia's coal country. Havard University, accessed January 24th 2024.

Kristrom, B. and P. Riera (1996). Is the income elasticity of environmental improvements less than one? *Environmental and Resource Economics 7*, 45–55.

Kumar, P., A. Kumar, P. Shinoj, and S. Raju (2011). Estimation of demand elasticity for food commodities in india. *Agricultural Economics Research Review 24*(1), 1–14.

Labandeira, X., J. M. Labeaga, and X. López-Otero (2017). A meta-analysis on the price elasticity of energy demand. *Energy policy 102*, 549–568.

Lee, S. and J. Lin (2018). Natural amenities, neighbourhood dynamics, and persistence in the spatial distribution of income. *The Review of Economic Studies 85*(1), 663–694.

Levinson, A. (2019). Energy efficiency standards are more regressive than energy taxes: Theory and evidence. *Journal of the Association of Environmental and Resource Economists 6*(S1), S7–S36.

Levinson, A. and J. O'Brien (2019). Environmental engel curves: Indirect emissions of common air pollutants. *Review of Economics and Statistics 101*(1), 121–133.

Li, S., J. Linn, and E. Muehlegger (2014). Gasoline taxes and consumer behavior. *American Economic Journal: Economic Policy 6*(4), 302–342.

Mardones, C. and C. Mena (2020). Economic, environmental and distributive analysis of the taxes to global and local air pollutants in chile. *Journal of Cleaner Production 259*, 120893.

Mayer, J., A. Dugan, G. Bachner, and K. W. Steininger (2021). Is carbon pricing regressive? insights from a recursive-dynamic cge analysis with heterogeneous households for austria. *Energy Economics 104*, 105661.

Metcalf, G. E. (1999). A distributional analysis of green tax reforms. *National Tax Journal*, 655–681.

Meya, J., S. Baumgärtner, M. A. Drupp, and M. F. Quaas (2020). Inequality and the value of public natural capital. *CESifo Working Paper* (No. 8752).

Meya, J. N. (2020). Environmental inequality and economic valuation. *Environmental and Resource Economics 76*(2), 235–270.

Mikkelson, G. M., A. Gonzalez, and G. D. Peterson (2007). Economic inequality predicts biodiversity loss. *PloS one 2*(5), e444.

Mildenberger, M., E. Lachapelle, K. Harrison, and I. Stadelmann-Steffen (2022). Limited impacts of carbon tax rebate programmes on public support for carbon pricing. *Nature Climate Change 12*(2), 141–147.

Mirrlees, J. A. (1971). An exploration in the theory of optimum income taxation. *The review of economic studies 38*(2), 175–208.

Muehlegger, E. and R. L. Sweeney (2022). Pass-through of own and rival cost shocks: Evidence from the us fracking boom. *Review of Economics and Statistics 104*(6), 1361–1369.

Murray, B. and N. Rivers (2015). British columbia's revenue-neutral carbon tax: A review of the latest "grand experiment" in environmental policy. *Energy Policy 86*, 674–683.

Neuhoff, K., S. Bach, J. Diekmann, M. Beznoska, and T. El-Laboudy (2013). Distributional effects of energy transition: impacts of renewable electricity support in germany. *Economics of Energy & Environmental Policy 2*(1), 41–54.

Nordhaus, W. (2019). Climate change: The ultimate challenge for economics. *American Economic Review 109*(6), 1991–2014.

Ohlendorf, N., M. Jakob, J. C. Minx, C. Schröder, and J. C. Steckel (2021). Distributional impacts of carbon pricing: A meta-analysis. *Environmental and Resource Economics 78*, 1–42.

Olson, M. (1965). *The Logic of Collective Action: Public Goods and the Theory of Groups, with a new preface and appendix*. Harvard University Press.

Ortiz-Bobea, A., T. R. Ault, C. M. Carrillo, R. G. Chambers, and D. B. Lobell (2021). Anthropogenic climate change has slowed global agricultural productivity growth. *Nature Climate Change 11*(4), 306–312.

Ostrom, E. (2009). A general framework for analyzing sustainability of social-ecological systems. *Science 325*(5939), 419–422.

Park, R. J., J. Goodman, M. Hurwitz, and J. Smith (2020). Heat and learning. *American Economic Journal: Economic Policy 12*(2), 306–39.

Park, R. J., N. Pankratz, A. P. Behrer, et al. (2021). Temperature, workplace safety, and labor market inequality. Technical report, Institute of Labor Economics (IZA).

Parry, I., H. Sigman, M. Walls, and R. Williams (2006). The incidence of pollution control policies. *The International Yearbook of Environmental and Resource Economics 2006/2007*, 1–42.

Parry, I. W. (2004). Are emissions permits regressive? *Journal of Environmental Economics and Management 47*(2), 364–387.

Parry, I. W. (2009). Pricing urban congestion. *Annu. Rev. Resour. Econ. 1*(1), 461–484.

Pizer, W. A. and S. Sexton (2019). The distributional impacts of energy taxes. *Review of Environmental Economics and Policy*.

Poterba, J. M. (1989). Lifetime incidence and the distributional burden of excise taxes. *The American Economic Review 79*(2), 325–330.

Preonas, L. (2023). Market power in coal shipping and implications for us climate policy. *Review of Economic Studies*, rdad090.

Rausch, S., G. E. Metcalf, and J. M. Reilly (2011). Distributional impacts of carbon pricing: A general equilibrium approach with micro-data for households. *Energy Economics 33*, S20–S33.

Rausch, S., G. E. Metcalf, J. M. Reilly, and S. Paltsev (2010). Distributional implications of alternative us greenhouse gas control measures. *The BE Journal of Economic Analysis & Policy 10*(2).

Rausch, S. and G. A. Schwarz (2016). Household heterogeneity, aggregation, and the distributional impacts of environmental taxes. *Journal of Public Economics 138*, 43–57.

Ravallion, M., M. Heil, and J. Jalan (2000). Carbon emissions and income inequality. *Oxford Economic Papers 52*(4), 651–669.

Sager, L. (2019). Income inequality and carbon consumption: Evidence from environmental engel curves. *Energy Economics 84*, 104507.

Sager, L. (2023). The global consumer incidence of carbon pricing: evidence from trade. *Energy Economics 127*, 107101.

Sandmo, A. (1975). Optimal taxation in the presence of externalities. *The Swedish Journal of Economics*, 86–98.

Schlenker, W. and M. J. Roberts (2009). Nonlinear temperature effects indicate severe damages to us crop yields under climate change. *Proceedings of the National Academy of Sciences 106*(37), 15594–15598.

Scruggs, L. A. (1998). Political and economic inequality and the environment. *Ecological economics 26*(3), 259–275.

Serebrisky, T., A. Gómez-Lobo, N. Estupiñán, and R. Muñoz-Raskin (2009). Affordability and subsidies in public urban transport: what do we mean, what can be done? *Transport reviews 29*(6), 715–739.

Smith, V. K. (2023). Accounting for income inequality in benefit transfers: The importance of the income elasticity of wtp. *Journal of Environmental Economics and Management 118*, 102781.

Somanathan, E., R. Somanathan, A. Sudarshan, and M. Tewari (2021). The impact of temperature on productivity and labor supply: Evidence from indian manufacturing. *Journal of Political Economy 129*(6), 1797–1827.

Spurlock, C. A. and K. S. Fujita (2022). Equity implications of market structure and appliance energy efficiency regulation. *Energy Policy 165*, 112943.

Steckel, J. C., I. I. Dorband, L. Montrone, H. Ward, L. Missbach, F. Hafner, M. Jakob, and S. Renner (2021). Distributional impacts of carbon pricing in developing asia. *Nature Sustainability 4*(11), 1005–1014.

Stern, N. (2007). *The economics of climate change: the Stern review*. Cambridge University press.

Sterner, T. (2012). *Fuel taxes and the poor: the distributional effects of gasoline taxation and their implications for climate policy*. Routledge.

Stiglitz, J. E. (2012). *The price of inequality: How today's divided society endangers our future*. WW Norton & Company.

Sun, C., M. E. Kahn, and S. Zheng (2017). Self-protection investment exacerbates air pollution exposure inequality in urban china. *Ecological Economics 131*, 468–474.

Tavoni, A., A. Dannenberg, G. Kallis, and A. Löschel (2011). Inequality, communication, and the avoidance of disastrous climate change in a public goods game. *Proceedings of the National Academy of Sciences 108*(29), 11825–11829.

Timmins, C. and A. Vissing (2022). Environmental justice and coasian bargaining: The role of race, ethnicity, and income in lease negotiations for shale gas. *Journal of Environmental Economics and Management 114*, 102657.

Torras, M. and J. K. Boyce (1998). Income, inequality, and pollution: a reassessment of the environmental kuznets curve. *Ecological economics 25*(2), 147–160.

Venmans, F. and B. Groom (2021). Social discounting, inequality aversion, and the environment. *Journal of Environmental Economics and Management*, 102479.

Vogt-Schilb, A., B. Walsh, K. Feng, L. Di Capua, Y. Liu, D. Zuluaga, M. Robles, and K. Hubaceck (2019). Cash transfers for pro-poor carbon taxes in latin america and the caribbean. *Nature Sustainability 2*(10), 941–948.

Walker, W. R. (2013). The transitional costs of sectoral reallocation: Evidence from the clean air act and the workforce. *The Quarterly journal of economics 128*(4), 1787–1835.

Wang, Q., K. Hubacek, K. Feng, Y.-M. Wei, and Q.-M. Liang (2016). Distributional effects of carbon taxation. *Applied energy 184*, 1123–1131.

Wang, X., G. Deltas, M. Khanna, and X. Bi (2021). Community pressure and the spatial redistribution of pollution: The relocation of toxic-releasing facilities. *Journal of the Association of Environmental and Resource Economists 8*(3), 577–616.

West, S. E. and R. C. Williams III (2004). Estimates from a consumer demand system: implications for the incidence of environmental taxes. *Journal of Environmental Economics and Management 47*(3), 535–558.

Weyl, E. G. and M. Fabinger (2013). Pass-through as an economic tool: Principles of incidence under imperfect competition. *Journal of Political Economy 121*(3), 528–583.

Wolverton, A. (2009). Effects of socio-economic and input-related factors on polluting plants' location decisions. *The BE Journal of Economic Analysis & Policy 9*(1).

Yip, C. M. (2018). On the labor market consequences of environmental taxes. *Journal of Environmental Economics and Management 89*, 136–152.

Yusuf, A. A. and B. P. Resosudarmo (2015). On the distributional impact of a carbon tax in developing countries: the case of indonesia. *Environmental Economics and Policy Studies 17*, 131–156.

Zhao, J. and L. Mattauch (2022). When standards have better distributional consequences than carbon taxes. *Journal of Environmental Economics and Management 116*, 102747.

Zivin, J. G., M. Neidell, and W. Schlenker (2011). Water quality violations and avoidance behavior: Evidence from bottled water consumption. *American Economic Review 101*(3), 448–453.

Online Appendix to: The Economics of Inequality and the Environment

Moritz A. Drupp[a,b], Ulrike Kornek[c,d], Jasper N. Meya[e,f], Lutz Sager[g]

[a] Department of Economics, University of Hamburg, Germany
[b] CESifo, Munich, Germany
[c] Department of Economics, Kiel University, Germany
[d] Mercator Research Institute on Global Commons and Climate Change, Germany
[e] Department of Economics, Leipzig University, Germany
[f] German Centre for Integrative Biodiversity Research (iDiv) Halle-Jena-Leipzig, Germany
[g] McCourt School of Public Policy, Georgetown University, USA

March 25, 2024

1 References to Table 1

[a] Alberini et al. (2011), [b] Zhou and Teng (2013), [c] Schulte and Heindl (2017), [d] Sun and Ouyang (2016), [e] Zhu et al. (2018), [f] Olmstead et al. (2007), [g] Cheesman et al. (2008), [h] Suárez-Varela (2020), [i] Basani et al. (2008), [j] Sebri (2014), [k] Blundell et al. (2012), [l] Díaz and Medlock (2021), [m] Wadud et al. (2010), [n] Sun and Ouyang (2016), [o] Labandeira et al. (2017), [p] Säll (2018)[q] ZHU et al. (2021), [r] Roosen et al. (2022), [s] Sheng et al. (2010), [t] Gallet (2010)

References

Alberini, A., W. Gans, and D. Velez-Lopez (2011). Residential consumption of gas and electricity in the us: The role of prices and income. *Energy Economics 33*(5), 870–881.

Basani, M., J. Isham, and B. Reilly (2008). The determinants of water connection and water consumption: Empirical evidence from a cambodian household survey. *World development 36*(5), 953–968.

Blundell, R., J. L. Horowitz, and M. Parey (2012). Measuring the price responsiveness of gasoline demand: Economic shape restrictions and nonparametric demand estimation. *Quantitative Economics 3*(1), 29–51.

Cheesman, J., J. Bennett, and T. V. H. Son (2008). Estimating household water demand using revealed and contingent behaviors: Evidence from vietnam. *Water resources research 44*(11).

Díaz, A. O. and K. B. Medlock (2021). Price elasticity of demand for fuels by income level in mexican households. *Energy Policy 151*, 112132.

Gallet, C. A. (2010). Meat meets meta: a quantitative review of the price elasticity of meat. *American Journal of Agricultural Economics 92*(1), 258–272.

Labandeira, X., J. M. Labeaga, and X. López-Otero (2017). A meta-analysis on the price elasticity of energy demand. *Energy policy 102*, 549–568.

Olmstead, S. M., W. M. Hanemann, and R. N. Stavins (2007). Water demand under alternative price structures. *Journal of Environmental economics and management 54*(2), 181–198.

Roosen, J., M. Staudigel, and S. Rahbauer (2022). Demand elasticities for fresh meat and welfare effects of meat taxes in germany. *Food Policy 106*, 102194.

Säll, S. (2018). Environmental food taxes and inequalities: Simulation of a meat tax in sweden. *Food Policy 74*, 147–153.

Schulte, I. and P. Heindl (2017). Price and income elasticities of residential energy demand in germany. *Energy Policy 102*, 512–528.

Sebri, M. (2014). A meta-analysis of residential water demand studies. *Environment, development and sustainability 16*, 499–520.

Sheng, T. Y., M. N. Shamsudin, Z. Mohamed, A. M. Abdullah, and A. Radam (2010). Demand analysis of meat in malaysia. *Journal of Food Products Marketing 16*(2), 199–211.

Suárez-Varela, M. (2020). Modeling residential water demand: An approach based on household demand systems. *Journal of environmental management 261*, 109921.

Sun, C. and X. Ouyang (2016). Price and expenditure elasticities of residential energy demand during urbanization: an empirical analysis based on the household-level survey data in china. *Energy Policy 88*, 56–63.

Wadud, Z., D. J. Graham, and R. B. Noland (2010). Gasoline demand with heterogeneity in household responses. *The Energy Journal 31*(1).

Zhou, S. and F. Teng (2013). Estimation of urban residential electricity demand in china using household survey data. *Energy policy 61*, 394–402.

ZHU, W.-b., Y.-f. CHEN, Z. Jing, and B.-b. WU (2021). Impacts of household income on beef at-home consumption: Evidence from urban china. *Journal of Integrative Agriculture 20*(6), 1701–1715.

Zhu, X., L. Li, K. Zhou, X. Zhang, and S. Yang (2018). A meta-analysis on the price elasticity and income elasticity of residential electricity demand. *Journal of Cleaner Production 201*, 169–177.